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# GENETIC VARIATION AND POPULATION STRUCTURE OF THE ENDANGERED SNAIL KITE IN SOUTH FLORIDA

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ABSTRACT.—Ten enzymatic stains were used to resolve the products of 12 loci for 150 snail kite (*Rostrhamus sociabilis*) nestlings from four major wetlands in south Florida. Nine loci were monomorphic across all sites; two loci were only slightly polymorphic, with overall allele frequencies <0.05. Average expected heterozygosity among all individuals was 4.6% (range = 0–25%). Average heterozygosity across the four sites ranged from 4.1–5.2%. Mean percent polymorphic loci (0.99 level) was 18.2% (range = 8.3–25%). Overall $F_{ST}$ was 3.4%, which was significantly different from 0; $F_{IS}$ and $F_{IT}$ values suggested a slight heterozygote deficiency. The largest genetic distance was consistently between Lake Okeechobee and the other sites; the shortest genetic distances were between Lake Kissimmee and Conservation Area 2B and between Conservation Area 2B and Conservation Area 3A. Gene flow was estimated at 7.1 migrants per generation. Short genetic distances among the four wetlands in south Florida suggest little differentiation among these populations of snail kites.

KEY WORDS: *snail kite*; Rostrhamus sociabilis; *Florida*; *electrophoresis*; *population genetics*.

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Variación genética y estructura poblacional de *Rostrhamus sociabilis* en peligro, en el sur de Florida

RESUMEN.—Diez colorantes enzimáticos fueron usados para analizar los productos de 12 loci para 150 polluelos de la especie *Rostrhamus sociabilis*, de los cuatro mayores humedales en el sur de Florida. Nueve loci fueron monomórficos a través de todos los sitios; dos loci fueron ligeramente polimórficos, con una frecuencia alélica total <0.05. El promedio esperado de heterozigocidad entre todos los individuos fue 4.6% (rango = 0–25%). El promedio de heterozigocidad a través de los cuatro sitios tuvo rangos entre 4.1–5.2%. La media porcentual de loci polimórficos (nivel 0.99) fue 18.2% (rango = 8.3–25%). $F_{ST}$ total fue 3.4%, significativamente diferente de 0; los valores de $F_{IS}$ y $F_{IT}$ sugieren un suave deficiencia heterozigotica. Consistentemente, la mayor distancia genética fue entre el Lago Okeechobee y los demás sitios; las menores distancias genéticas se registraron entre el Lago Kissimmee y el Area de Conservación 2B y entre el Area de Conservación 2B y el Area de Conservación 3A. El flujo genético fue estimado en 7.1 migrantes por generación. Las pequeñas distancias genéticas entre los cuatro humedales del sur de Florida sugieren poca diferenciación entre las poblaciones de *R. sociabilis*.

[Traducción de Ivan Lazo]

The snail kite (*Rostrhamus sociabilis*) occurs widely in tropical Central and South America, Cuba and Florida (Sykes et al. 1995). *R. s. plumbeus* is restricted to Cuba and southern Florida. Movement between Florida and Cuba is doubtful given

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the lack of foraging habitat in extreme south Florida (e.g., Florida Bay and Keys), the short-distance nomadic dispersal shown in Florida and the relatively large expanse of open water separating Florida and Cuba. The original breeding range in Florida primarily consisted of the headwaters of the St. Johns River northward to the Oklawaha drainage, the Kissimmee River basin (including Lake Kissimmee), southward through Lake Okeechobee, the Everglades and freshwater marshes near Florida Bay (Sykes 1984). However, by the late 1960s Sykes (1984) found kites mostly at Lake Okeechobee, Conservation Area 1 (Loxahatchee NWR), Conservation Area 2A, Conservation Area 2B (CA2B) and the southern portion of Conservation Area 3A (CA3A). These conservation areas are large impounded remnants of the Everglades habitat that once extended from Lake Okeechobee to the northern edge of Everglades National Park. The range of snail kites became further reduced to the marshes on the west side of Lake Okeechobee and the southern region of CA2B and CA3A during the 1970s (Sykes 1984). The range decline in Florida along with the large-scale decrease in numbers of kites resulted in the species being listed as endangered on the initial federal endangered species list in 1967 (Fed. Reg. 42[155]:40685–40688). It was similarly listed as endangered on the initial state of Florida list in 1972.

Considerable inter-year variation has occurred in the numbers of snail kites found at individual wetlands in Florida during the 1970s and 1980s (Beissinger and Takekawa 1983, Rodgers et al. 1988, Takekawa and Beissinger 1989, Bennetts et al. 1994). These fluctuations often were associated with low water levels and droughts that force the birds to disperse to other wetlands. Based on nest monitoring and sightings of color-banded birds, kites dispersed from the southern parts of their range and recolonized their former nesting range at Lake Kissimmee, Lake Tohopekaliga, East Lake Tohopekaliga, the upper St. Johns River marshes in Indian River County and several smaller wetlands in Hendry and Okeechobee counties during a particularly severe drought in the late 1980s (Sykes et al. 1995). Apparently, lack of suitable foraging habitat and decreased availability of apple snails (*Pomacea paludosa*), precludes recolonization farther north.

Because of these recent fluctuations both in the size and range of this relict population within Florida, the snail kite warrants a genetics study to determine if it has experienced loss of genetic variability due to population bottlenecks. The objectives of our study were therefore to (1) document the level of genetic variability in populations nesting in four major wetlands in south Florida and (2) estimate levels of genetic differentiation among these populations. This information would provide insight into the effects of dispersal on population genetics and allow management decisions to be made regarding snail kite recovery in the state of Florida.

## METHODS

Our study was conducted under the Florida Administrative Code, General Purpose Wildlife Code 39-9.002, subsection 2, that permits Florida Game and Fresh Water Fish Commission personnel and cooperating investigators to handle birds for specific purposes of approved research. Feather tissues also were collected under the authority of the Endangered Species Cooperative Agreement between the U.S. Fish and Wildlife Service and the Florida Game and Fresh Water Fish Commission. Our field work followed the American Ornithologists' Union guidelines for scientists conducting research on wild birds (Oring et al. 1988).

We collected tissue samples from snail kite nestlings in the four major wetlands (Lake Kissimmee, Lake Okeechobee, CA2B, CA3A) of the species' range in south Florida during 1987 (Fig. 1). One growing, centrally-located secondary feather from each wing was removed from one nestling (3–4 wk of age) per nest. Feathers were frozen in liquid nitrogen within 1 min after removal and subsequently stored in an ultra-cold freezer (−76°C) until electrophoresed. Laboratory (Stangel 1986) and field (Stangel and Lennartz 1988) studies indicate that feather removal is not detrimental to survival or growth of even small birds.

Pulp was squeezed from the feather shafts and homogenized with 5 ml of 0.01 M Tris-0.001 M EDTA pH 7.0 buffer solution. Electrophoretic conditions and general staining procedures followed those techniques of Selander et al. (1971), Harris and Hopkinson (1976) and Barman (1985). Loci were numbered according to the mobility of their products from anode to cathode. Allozymes were designated alphabetically in order of relative mobility from anode to cathode, with the letter "C" chosen to represent the most common allele.

The statistical package BIOSYS-1 (Swofford and Selander 1981) was used for analysis of snail kite allelic frequencies, genetic variability measures (Hardy-Weinberg expected heterozygosity, mean number of alleles per locus, percent polymorphic loci), deviations from expected Hardy-Weinberg proportions, Nei's (1978) and Rogers' (1972) genetic distances and F-statistics (Nei 1977, Wright 1978). Gene flow, or Nm (N = deme size and m = migration rate among demes), was calculated using Wright's (1943) formula: $F_{ST} = 1/(4Nm + 1)$.

Patterns of population structure can be revealed through analysis of allele frequencies using Wright's F-statistics (Wright 1978). The most commonly used statistic, $F_{ST}$, is a measure of the extent that a species shows spatial

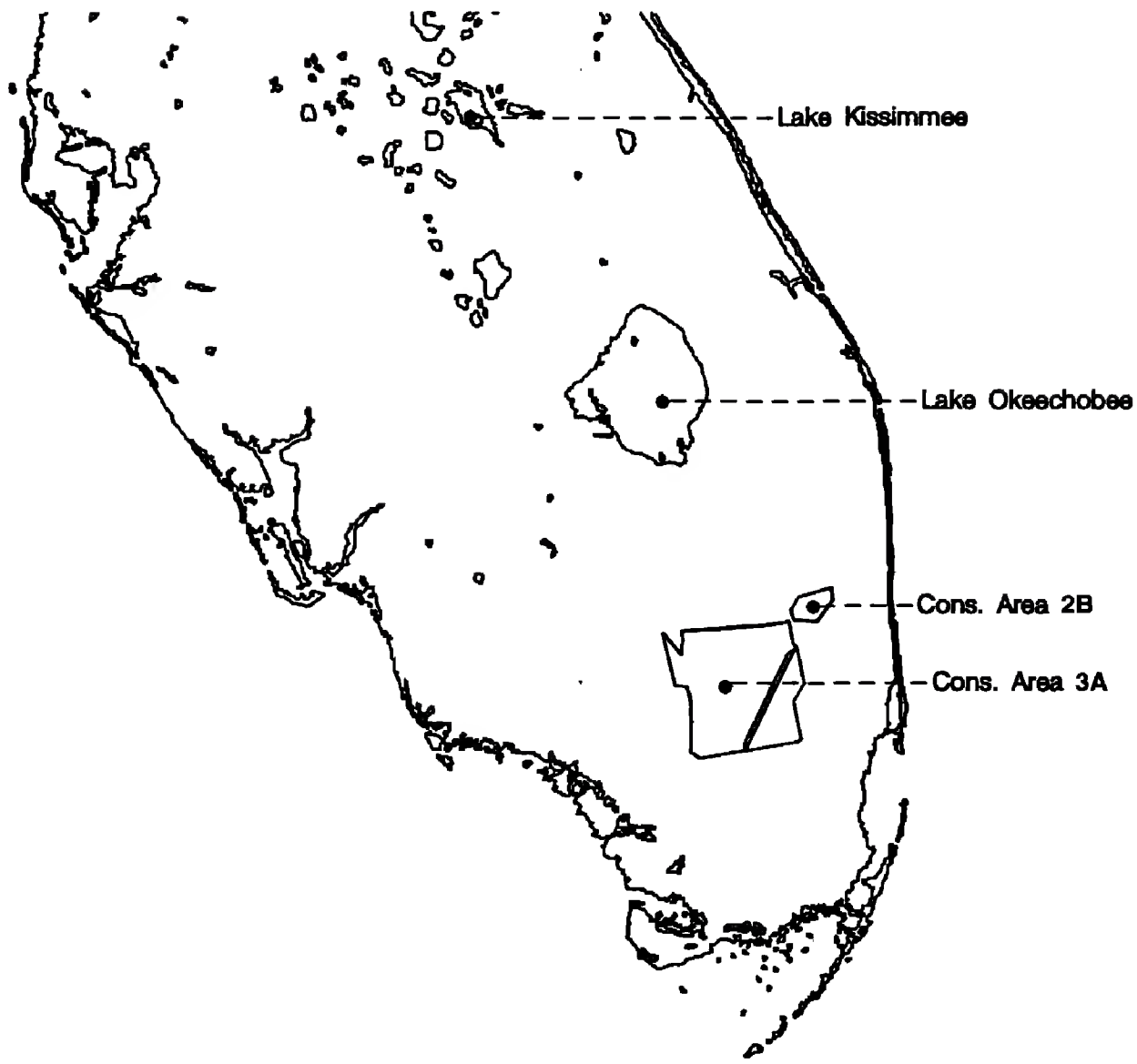


Figure 1. Sources of tissue samples from snail kite nestlings at four wetlands in south Florida.

genetic heterogeneity. $F_{ST}$ values range from 0, suggesting lack of differentiation or panmixia, to 1, indicating fixation of alternative alleles and complete differentiation. $F_{IS}$ and $F_{IT}$ are measures of heterozygote deficiency or excess within subpopulations (e.g., the four wetlands in our study) and the total population, respectively, and are commonly used as inbreeding indices. Both values range from $-1$ to 1, with positive values indicating heterozygote deficiency, which may occur with inbreeding. Precise interpretation of F-statistics requires detailed knowledge about the breeding structure of the species examined. Because this information is lacking in the snail kite, inferences about population structure cannot be made unambiguously.

## RESULTS

Ten enzymatic stains were used to resolve the products of 12 loci for 150 snail kite nestlings from the four wetland sites. All individuals were scored for all loci. Nine loci were monomorphic across all sites; two loci were only slightly polymorphic (Pgd and Tri) and a single locus (Pep) exhibited three alleles, with overall allele frequencies $<0.05$ (Appendix 1). There was no convincing evidence for significant deviation from Hardy-Weinberg proportions.

Average expected heterozygosity among all individuals was 4.6% and ranged from 0–25%; the most heterozygous individual was therefore variable at 3 of 12 loci. Fifty-two percent of all individuals were monomorphic, 43% were heterozygous at one locus, 4% were heterozygous at two loci and one individual was heterozygous at three loci. Average heterozygosity across the four sites ranged from 4.1–5.2% (Table 1). Percent polymorphic loci (0.99 level) ranged from 8.3–25%, with a species mean of 18.2%. Mean number of alleles per locus ranged from 1.1–1.3, with a mean of 1.2.

Table 1. Genetic variability at 12 loci for snail kites from four wetlands in south Florida. Standard errors are in parentheses.

| SITE | *N* | MEAN NUMBER OF ALLELES PER LOCUS | PERCENTAGE OF LOCI POLYMORPHIC (0.99) | MEAN HETEROZYGOSITY HARDY-WEINBERG EXPECTED (×100) |
|---|---|---|---|---|
| Kissimmee | 24 | 1.1 (0.1) | 8.3 | 4.2 (4.2) |
| Okeechobee | 24 | 1.3 (0.1) | 25.0 | 5.2 (3.7) |
| Cons. Area 2B | 26 | 1.2 (0.1) | 16.7 | 4.2 (3.8) |
| Cons. Area 3A | 76 | 1.1 (0.1) | 8.3 | 4.1 (4.1) |

Table 2. F-statistics for three polymorphic loci in snail kites from south Florida.

| LOCUS | $F_{IS}$ | $F_{IT}$ | $F_{ST}$ |
|---|---|---|---|
| Phosophogluconate dehydrogenase (Pgd) | 0.064 | 0.096 | 0.035 |
| Peptidase phynylproline (Pep) | −0.046 | −0.015 | 0.030 |
| Tripeptide aminopeptidase (Tri) | −0.041 | −0.010 | 0.030 |
| Mean | 0.053 | 0.085 | 0.034 |

The frequency of the common allele ("C") at the Pgd locus ranged from 0.438–0.667 (Appendix 1), which was significantly different from 0 among sites ($\chi^2_3 = 8.4$, $P = 0.037$). Three private polymorphisms (alleles detected at only one site) were identified: Pep "D" allele at CA3A; Pep "B" allele at CA2B; and Tri "D" allele at CA3A.

An overall $F_{ST}$ of 3.4% was significantly different from 0 ($\chi^2_{16} = 37.5$, $P < 0.005$; Table 2). $F_{IS}$ and $F_{IT}$ values were on average positive, suggesting a slight heterozygote deficiency. Mean Nei's genetic distance among sites was 0.002 (range = 0.000–0.004). Mean Rogers' genetic distance among sites was 0.014 (range = 0.006–0.027). The largest genetic distances using both methods were consistently between Lake Okeechobee and the other sites (Nei's distance: 0.002–0.004; Rogers' distance: 0.014–0.027), whereas, the shortest genetic distances were between Lake Kissimmee and CA2B (Nei's distance: 0.000; Rogers' distance: 0.006) and between CA2B and CA3A (Nei's distance: 0.000; Rogers' distance: 0.008). Gene flow, or the estimated number of migrants per generation, was 7.1.

## DISCUSSION

Heterozygosity in the snail kite (4.6%) was slightly lower than the average of 6.5% (range = 0–30.7%) for 86 species of birds reported by Evans (1987). However, average percent polymorphic loci (18.2) and alleles per locus (1.2) of kites were within the range of those reported for other bird species (Barrowclough et al. 1985, Evans 1987). Because we lack data for comparisons with populations outside of Florida and historical populations of snail kites in the state, we do not know if the current level of heterozygosity in Florida is similar for the species over its entire range, or differs because of founder effects when the species originally colonized Florida and/or population bottlenecks experienced during population decreases that occurred in the 1960s.

Reported population fluctuations in the snail kite (Beissinger and Takekawa 1983, Rodgers et al. 1988, Takekawa and Beissinger 1989, Bennetts et al. 1994) might have affected heterozygosity, although population bottlenecks would have to be severe and last several generations to have a significant effect (Nei et al. 1975). The presence of three private polymorphisms suggests that these conditions have not occurred and the historical reduction in the south Florida population does not seem to have affected levels of genetic variation. Heterozygosity in the snail kite varied from 4.1% to 5.2% at the population level, which is within typical values for birds (Evans 1987).

Genetic differentiation among snail kites in the four wetlands we sampled was low, as it is for most species of birds (Evans 1987). An $F_{ST}$ of 0.034 in the snail kite suggests that only about 3.4% of the total genetic variation detected can be accounted for by heterogeneity among sites, with the remaining 96.6% accounted for within sites. Evans (1987) reported an average of 4.8% of the variation occurs among populations for 23 avian species he examined. Short genetic distances among the four wetlands in south Florida also suggest little differentiation among these populations of snail kites.

Although habitat fragmentation tends to contribute to smaller, more isolated populations, the ability of birds to fly would be expected to increase gene flow and contribute to lack of genetic differentiation among even distant sites. Periodic population shifts by snail kites from the Everglades conservation areas caused by changing hydrologic con-

ditions likely increases gene flow among the south Florida wetlands used as breeding sites. Periodic low water levels that cause kites to disperse can result in low recruitment, increased adult mortality and population decreases (Beissinger 1995). Kites also exhibit extensive annual movements that seem to have nothing to do with hydrologic conditions (Bennetts and Kitchens 1992, 1993). The estimated 7.1 migrants per generation calculated from $F_{ST}$ using Wright's (1943) formula further suggests high levels of interchange among south Florida demes. Ideally, the exchange of animals should be pulsed and occur at times when inbreeding has become great enough such that outbreeding will yield optimum levels of heterosis (Chesser et al. 1980). Thus, even with the snail kite population subdivided among south Florida wetlands, the stochastic nature of drought events should contribute to pulsed exchange of individuals that would be able to maintain heterozygosity and a vigorous population.

That the snail kites at Lake Okeechobee were slightly more genetically distant than kites at the other three wetlands is interesting, but the genetic distances are very short and must be interpreted cautiously. Kites were rarely observed and did not breed at Lake Kissimmee during the 1960s and 1970s (Sykes 1984). Kites began to breed at Lake Kissimmee during the early 1980s when drought conditions forced them to abandon the Everglades (e.g., CA2B and CA3A). The reason why kites from the Everglades would pass Lake Okeechobee and move farther north to Lake Kissimmee is unclear. Perhaps a drawdown of the lake in 1979 for fisheries management restored suitable foraging conditions that facilitated the recolonizing of Lake Kissimmee. However, until more is known about the response of apple snails to another drawdown of the lake during 1996, we are reluctant to speculate further. Recolonization also may have been due to the large number of kites that already occupied Lake Okeechobee and concurrent low lake levels during the early 1980s. Perhaps the breeding subpopulation at Lake Okeechobee is more stable than those at the other three major wetlands. Whereas lake levels, amount of flooded marsh and number of kites vary among years (Rodgers 1992), some littoral zone at Lake Okeechobee always is available for foraging and nesting. If the other wetlands flood and dry out as the result of frequent, extreme hydrological changes, there may be considerable exchange of kites among these sites that increases their genetic similarity relative to Lake Okeechobee. Zink et al. (1987) also found more similarity between farthest separated populations of California Quail (*Callipepla californica*) than nearest geographic neighbors.

The logical next step for genetic studies of the snail kite would be to sample populations in Cuba and South and Central America where the species is common and widespread (Beissinger 1983, Beissinger et al. 1983). Comparisons with the south Florida population would provide insight into genetic differentiation and hence, gene flow between these sites. It also would be interesting to compare these areas to see if the south Florida population exhibits reduced genetic variability relative to the larger and potentially more stable and representative South American populations.

### CONSERVATION IMPLICATIONS

The results of this study provide benchmark snail kite genetic variability measures against which values obtained in the future can be compared. If future population fluctuations reduce snail kite numbers to very low levels, it would then be possible to determine if genetic variability had been reduced relative to our values.

Translocation of individuals from larger populations is one strategy to increase small populations but consideration must be given to the genetic characteristics of both donor and recipient populations to lessen the chance of disrupting locally adapted populations (Avise and Nelson 1989, Stangel et al. 1992). Although translocation of kites has not been considered to date, our data can serve as a reference for managers considering translocation of genetically similar snail kites to south Florida if ever conditions warrant such a drastic recovery strategy.

With the exception of the Lake Okeechobee subpopulation, we found few distinctions among the snail kite demes in south Florida. The Lake Okeechobee subpopulation should receive further study, particularly with regard to the movement of successfully breeding kites into and out of this wetland relative to other wetlands. Kites have exhibited a tendency to concentrate their population in CA3A during some years (Sykes 1984, Rodgers et al. 1988, Bennetts et al. 1994, Sykes et al. 1995). A large number of an endangered species at a single site is a poor conservation strategy from a genetic point of view. However, extensive annual movements by kites often result in considerable inter-

change of birds among wetlands in south Florida (Bennetts and Kitchens 1992, 1993). The challenge will be to maintain a continued interchange of individuals among these sites for a high degree of genetic polymorphism while at the same time minimizing the effects of inbreeding within each wetland in south Florida.

The population and distribution fluctuations of snail kites in south Florida are so dramatic that demographic concerns probably outweigh immediate genetic threats and these should receive greatest attention in conservation plans. Appropriate demographic and habitat management of the snail kite will prevent the loss of genetic variability due to population bottlenecks.

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Appendix 1. Allele frequencies and electrophoretic conditions for three polymorphic loci of snail kites from four wetlands in south Florida.[a]

| Enzyme Locus Allele | Enzyme Commission Number[b] | Wetland | | | |
|---|---|---|---|---|---|
| | | Kissimmee | Okeechobee | Cons. Area 2B | Cons. Area 3A |
| Phosphogluconic dehydrogenase (Pgd) | 1.1.1.43 | | | | |
| C | | 0.604 | 0.438 | 0.654 | 0.667 |
| D | | 0.396 | 0.563 | 0.346 | 0.333 |
| Phenylalanyl-proline peptidase (Pep) | 3.4.13.9 | | | | |
| B | | 0.000 | 0.000 | 0.019 | 0.000 |
| C | | 1.000 | 1.000 | 0.981 | 0.947 |
| D | | 0.000 | 0.000 | 0.000 | 0.053 |
| Tripeptide aminopeptidase[c] (Tri) | 3.4.1.4 | | | | |
| C | | 1.000 | 1.000 | 1.000 | 0.961 |
| D | | 0.000 | 0.000 | 0.000 | 0.039 |

[a] The following 9 loci were monomorphic in all individuals assayed: malic dehydrogenase-1 and malic dehydrogenase-2 (1.1.1.37); lactate dehydrogenase-1 (1.1.1.27); phosphogluco-isomerase (5.3.1.8); creatinine kinase-1 and creatinine kinase-2 (2.7.3.2); isocitrate dehydrogenase-1 (1.1.1.42); phosphoglucomutase (5.4.2.2); leucine aminopeptidase (3.4.11).

[b] Enzyme commission number from Barman (1985).

[c] Substrate for tripeptide aminopeptidase was leucylglycylglycine.

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# USE OF DNA ANALYSIS TO IDENTIFY SEX OF NORTHERN SPOTTED OWLS (*STRIX OCCIDENTALIS CAURINA*)

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ABSTRACT.—The spotted owl (*Strix occidentalis*) is a monochromatic species with slight sexual size dimorphism in adults. Methods currently available to identify sex of adult owls are ineffective for juveniles. Blood samples taken from owls from the eastern Cascade Mountains, Washington were used for cDNA cloning of a Z- and W-linked gene, DZWM1, to identify the sex of adult owls of known sex. A blind assessment resulted in the correct identification of sex for all 59 owls sampled (45 subadult/adult and 14 juveniles recaptured as subadult/adult). We believe this technique can be used to identify the sex of owls that cannot otherwise be identified using less invasive morphometric methods. Both field and laboratory procedures are described.

KEY WORDS: *DNA*; *sex identification*; *spotted owl*; Strix occidentalis.

Uso de análisis de DNA para identificar sexo en *Strix occidentalis caurina*

RESUMEN.—*Strix occidentalis* es una especie monocromática con escaso dimorfismo sexual en el tamaño adulto. Los métodos comunmente disponibles para identificar sexo en los búhos adultos no son efectivos para juveniles. Muestras de sangre obtenidas de búhos del este de las Montañas Cascada, Washington, fueron usadas para clonación de cDNA de un gen Z-y W-ligado, DZWM1, para identificar el sexo de individuos ya determinados. Estas medidas resultaron en la identificación correcta del sexo para los 59 búhos muestreados (45 subadultos/adultos y 14 juveniles recapturados como subadultos/adultos). Creemos que esta técnica puede ser usada para identificar el sexo de búhos que, de otra manera, no podrían ser identificados usando métodos morfométricos menos invasivos. Tanto el procedimiento de campo como el de laboratorio son descritos.

[Traducción de Ivan Lazo]

The spotted owl (*Strix occidentalis*) is a monochromatic species with slight sexual size dimorphism in adults (Blakesley et al. 1990). Morphometric models and behavioral clues, such as vocalizations, have been used to identify the sex of adult spotted owls (Forsman et al. 1984, Blakesley et al. 1990). However, none of these methods are useful in identifying sexes of juvenile birds. It may be important to correctly sex juvenile spotted owls because the survival rate of juveniles is a significant element of models used to estimate the status of populations (Burnham et al. 1994). For example, information on habitats used by juveniles during dispersal can provide data on the effects of landscape conditions on age- and/or sex-specific survival.

Both hormone immunoassay and genetic analysis have been used to identify sex of birds. Two steroid hormone immunoassay methods have been developed and used for a number of species, but the level of accuracy varies among species (Bercovitz et al. 1978, Tell and Lasley 1991). Neither has

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been used in field conditions where it would be difficult to obtain multiple fecal samples from the same individual.

Genetic methods of sex identification such as chromosome analysis (karyotyping) and flow cytometry have been used with varying levels of success (Ivins 1975, Halverson et al. 1985, Prus and Schmutz 1987, Tiersch and Mumme 1993). Only Nakamura et al. (1990) and Valentine (1990) have reported high (99–100%) rates of correct sexing, using chromosome analysis and flow cytometry, respectively. In addition, analysis of nucleotide sequences of differential regions of avian sex chromosomes generally has only limited usefulness in diverse avian groups (Uryu et al. 1989, Quinn et al. 1990, de Kloet and de Kloet 1991, Longmire et al. 1991). Microsatellite probes can also be used to determine gender in a wide variety of avian species (Longmire et al. 1993).

Recently, cDNA cloning of a Z- and W-linked gene, DZWM1, from the domestic turkey (*Meleagris gallopavo*) has proven a reliable method to identify sex in psittacines and it has been increasingly tested successfully on many avian orders (Halverson 1990, Dvorak et al. 1992). Reliable procedures need to be developed for many species (Griffiths and Tiwari 1993, J. Longmire pers. comm.). Probes of this gene sequence are homologous to the differential region of both sex chromosomes and sex can be determined by restriction fragment length polymorphism.

The objective of this study was to determine the accuracy and utility of using cDNA cloning techniques to identify the sex of spotted owls. Our goal was to use this technique to verify the sex of juvenile owls to facilitate development of a noninvasive morphometric model for identifying the sex of juveniles under field conditions. Herein, we describe collection and analysis techniques and our results using DZWM1 to identify the sex of spotted owls.

## METHODS

We collected blood samples from 45 subadult or adult spotted owls captured in the eastern Cascade Mountains of Washington in 1991–95. In addition, blood samples were collected from 308 juveniles, of which 14 were recaptured as subadults/adults on the study area. Owls were captured using techniques described by Forsman (1983); criteria for identifying sex of subadult and adult spotted owls are reported by Blakesley et al. (1990). Most blood samples were taken from members of pairs (19 of 20 females, 22 of 25 males).

Blood was drawn from the brachial vein of the wing after swabbing the area with alcohol disinfectant. Approximately 20 μl of blood was drawn using a 1.0 cc tuberculin syringe with a 22–25 ga needle. In 1991–92 the blood was flushed directly into a cryovial containing 70% ethanol to prevent contamination. In 1993–94, a heparinized capillary tube was used to collect 0.1–0.2 μl of blood. This technique did not involve direct withdrawal of blood. Rather, the vein was punctured and several drops of blood that accumulated on the surface of the skin were drawn into the capillary tube and then placed in the cryovial. Samples were labeled, refrigerated at 7°C for 2–10 d, then sent by mail to the laboratory for processing and identification of sex without information on the field identification of the owl.

Samples were centrifuged at 2000 rpm for 5 min in the initial collection tube. The supernatant was discarded and the residual pellet resuspended in DNA Isolation Buffer (0.1 M sodium chloride [NaCl], 0.05 M Tris pH 8, 0.1 M EDTA with 0.2 mg/ml Proteinase K and 0.5% SDS). The tubes were gently rocked at 55°C for 1–4 hr.

Following incubation, 0.5 μl saturated NaCl solution was added and the samples shaken vigorously for 15 sec. The samples were then centrifuged at 2000 rpm for 15 min and the supernatant decanted into a fresh tube, discarding the pellet. Three ml of 95% ethanol were added to the supernatant, and the solution gently mixed until the DNA precipitate formed. The precipitate was removed using a glass pipette hook, rinsed in 70% ethanol and allowed to air dry. The DNA was resuspended in TE (10 mM Tris pH 8, 2 mM EDTA) and dissolved by gentle rocking at 55°C for 1 d.

Twenty-five μl of sample (approximately 5 μg) was digested with SacI according to manufacturer's recommendations (Pharmacia Biotec, Piscataway, NJ, U.S.A.) in a total reaction volume of 35 μl. Ten μl of "Stop Dye" (Maniatis et al. 1982) was added and the samples loaded on a 0.8% agarose TBE gel. Electrophoresis was performed at 15°C with recirculating buffer at 45 amperes for 16–18 hr.

After electrophoresis, gels were photographed and trimmed. Gels were immersed in denaturant (1.5 M NaCl, 0.5 NaOH) for 30 min with gentle agitation, rinsed briefly with water, then immersed in neutralizer (1 M Tris pH 7.4, 1.5 M NaCl) for 30 min with gentle agitation. Gels were then set up for capillary blotting onto Hybond N (Amersham Corporation, 2636 South Clearbrook Drive, Arlington Heights, IL, U.S.A. 60005) with 10× SSC as the transfer buffer. Transfer was complete after 8–16 hr.

Blots were removed from the blotting apparatus, rinsed briefly in 5× SSC, and air dried. Blots were exposed for 2 min on a UV transilluminator to crosslink the DNA. Blots were then moistened in prehybridization fluid (0.5 M NaPhos pH 7.0, 7% SDS, 5 mM EDTA) and loaded into glass tubes with 5 ml of prehybridization fluid. The tubes were heated in the incubator for at least 15 min. The probe was added and hybridization proceeded for 12–18 hr at 65°C.

DNA analysis was performed as previously reported (Dvorak et al. 1992) except that restriction digests were performed with SacI.

Blots were washed twice in 2× SSC and 0.5% SDS for 15 min each at room temperature, then washed in 0.2× SSC, 0.5% SDS, for 25 min at 52°C. Blots were exposed

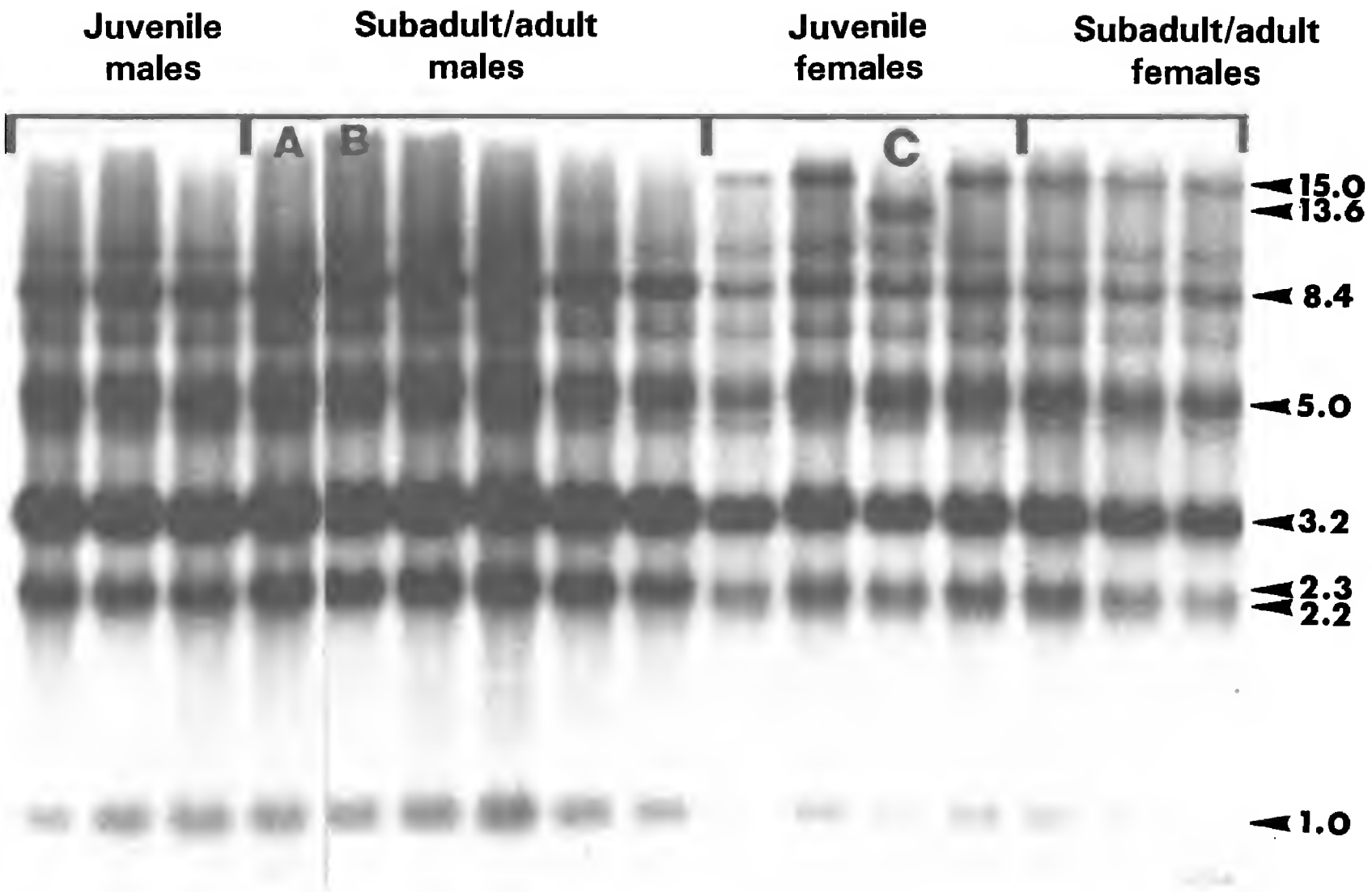


Figure 1. SacI digests of 16 northern spotted owls probed with DZWM1. Lanes A and B are samples from the same bird taken one year apart (1991 and 1992). Lane C shows an uncommon W allele in this study.

to Fuji RX film with two intensifying screens. Overnight exposure was generally sufficient.

## RESULTS

We correctly identified the sex of all 45 subadult/adult spotted owls from which sufficient DNA had been collected. Three birds (one male [Fig. 1], two females) were sampled twice, once as fledglings and again as first-year subadults. We also observed three second-year owls (two males, one female; not included in totals above) which, as juveniles, were correctly identified by other researchers who used this same procedure (D. Herter pers. comm., S. Sovern pers. comm.).

An autoradiograph of Southern blot hybridization of DZWM1 on SacI digest of genomic DNA from 16 spotted owls is shown in Fig. 1. The arrows on the right side of the figure indicate the molecular weight size of the bands in kilobases (kb). With SacI digestion, the 15.0 and 13.6 kb bands are found only in female birds, hence are derived from the differentiated region of the W chromosome. The 8.4, 3.2, 2.3 and 1.0 kb bands, which are of double-intensity in male birds, originate from the Z chromosome. The 13.6 kb band found in sample C is an alternate W allele. Though comparatively rare in our Washington samples, it is more common in samples from Oregon (Forsman pers. comm., Halverson pers. obs.). The 5.0 kb band is nonspecific and provides a crude measure of the amount of DNA in each lane.

## DISCUSSION

DNA analysis correctly identified the sex of all 62 (45 subadult/adult, three resamples, 14 recaptures) samples of spotted owl blood. This technique can be used by commercial laboratories and has immediate application in studies of spotted owl ecology because it allows rapid sex identification of all juvenile owls and eliminates reliance on recaptures to determine sex (the recapture rate of this sample was only 3.1%; Fleming unpubl. data). Because noninvasive sex determination models based on morphometric features appear capable of accurately sexing most spotted owls (Fleming unpubl. data), we recommend blood sampling only those owls which cannot be reliably sexed by measurement.

Analyses involving other avian species indicate that blood sampling procedures have little effect on individual birds (Stangel 1986, Colwell et al. 1988, Stangel and Lennartz 1988, Ardern et al. 1994). Nonetheless, we attempted to evaluate whether spotted owls were adversely effected by handling and blood sampling. Our resighting rate for adults from which we collected blood samples was 88.9% (40/45). Of the five owls that have not been resighted, two were from areas burned by a catastrophic forest fire in 1994, and there have been no opportunities to resight three owls banded in 1995. It would be difficult to make a similar assessment for juveniles because the resighting rate is very low. However, extensive observations on two adjacent study areas of radio-equipped juveniles from which blood samples were collected indicate that blood sampling does not adversely effect the owls (D. Herter pers. comm., S. Sovern pers. comm.). This leads us to believe that the procedure is safe when properly administered. Because the northern spotted owl is a threatened subspecies, we recommend that all blood samples be collected under required Bird Banding Laboratory special authorization by banders with experience and/or training in blood extraction procedures.

We encountered very few problems using this procedure. Two samples, correctly identified in the laboratory, were initially misidentified when a technician inadvertently trimmed the photographs of the blots too closely. Only 20 of 358 samples (355 birds and three samples that were tested twice), yielded no initial results. Of these, six were misplaced; three contained marginal amounts of blood, but were salvaged when results were obtained on a retest; four contained insufficient blood to analyze; and seven apparently degraded due to heat exposure when transported from the field at temperatures of about 37°C. Heat degradation had not been previously observed in over 50 000 samples from 300 avian species. Because spotted owl blood coagulated rapidly in hot weather (≥29°C) upon exposure to air or to a needle exposed to the sun, we often placed blood directly into the cryovial from the syringe. This occasionally necessitated additional laboratory time to eliminate "extra" sample. The problem was corrected when we later used a heparinized capillary tube to collect blood.

Sex-linked polymorphism exhibited by Southern blot hybridization with DZWM1 is consistently found with other restriction enzymes; the choice of enzyme depends on ease of interpretation and cost. Digestion with alternate enzymes can aid in the analysis of allelic variation such as occurred in sample C. The existence of variant alleles necessitates the development of a data base on a given species before accurate sexing of unknown samples can be guaranteed. We therefore recommend that initial efforts use, if possible, blind samples from known-sex members of pairs to establish species parameters.

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# BOREAL OWL MATING HABITAT IN THE NORTHWESTERN UNITED STATES

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ABSTRACT.—We examined boreal owl (*Aegolius funereus*) mating habitat in the Sierra Madre range of the Medicine Bow National Forest in Wyoming in the northwestern United States during 1992–93. In nocturnal surveys, we found 22 boreal owl singing locations which we compared to 68 random locations in the study area. Owls used stands dominated by Engelmann spruce (*Picea engelmannii*) and subalpine fir (*Abies lasiocarpa*) more often (77%) than stands of lodgepole pine (*Pinus contorta*). Stand size ranged from 0.2–122.8 ha though adjacent stand type (forest or opening) was not investigated. All boreal owls were found in areas with old forest characteristics including high basal areas of trees, tall snags, many large down logs, and a tall overstory canopy. Some boreal owls sang from old forest stands adjacent to clearcuts, one as close as 15 m. Older forests may provide nest holes for boreal owls which are obligate cavity nesters.

KEY WORDS: *boreal owl*; Aegolius funereus; *mating habitat*; *old growth*; *Sierra Madre Forest*; *clearcut.*

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Habitat de apareamiento de *Aegolius funereus* en el noroeste de los Estados Unidos

RESUMEN.—Durante 1992 y 1993, examinamos el hábitat de apareamiento de *Aegolius funereus* en la Cordillera de Sierra Madre del Medicine Bow National Forest en Wyoming, al noroeste de los Estados Unidos. En recorridos nocturnos, encontramos 22 sitios de canto de este búho, los que comparamos con 68 sitios al azar en el área de estudio. Los búhos usaron más a menudo (77%) parches dominados por *Picea engelmannii* y *Abies lasiocarpa* que parches de *Pinus contorta.* El rango del tamaño del parche fue de 0.2 a 122.8 ha. Todos los búhos fueron encontrados en áreas con características de bosque antiguo, incluyendo grandes áreas basales de árboles, ramas espigadas, grandes troncos caídos y grandes sotobosques. Algunos individuos de esta especie se encuentran hasta alrededor de 15 m de claros de bosque. Antiguos bosques pueden proveer de sitios de nidificación para estos búhos que nidifican obligadamente en cavidades.

[Traducción de Ivan Lazo]

Boreal owl (*Aegolius funereus*) mating and nesting habitat is poorly understood in the U.S. because of their nocturnal behavior in remote forests. During their late winter mating season in the western states, boreal owls associate with older high-elevation forests generally composed of Engelmann spruce (*Picea engelmannii,* ES) and subalpine fir (*Abies lasiocarpa,* SA) (Webb 1982, Palmer 1986, Hayward et al. 1993). High elevation patches of ponderosa pine (*Pinus ponderosa*), Douglas fir (*Pseudotsuga menziesii*), and quaking aspen (*Populus tremuloides*) are important when available (Hayward et al. 1993). These forest types also support the owl's primary prey, the southern red-backed vole (*Clethrionomys gapperi*) (Hayward et al. 1993).

During the mating season, male owls sing for a mate with a continuous high-pitched song that can easily be heard from 1.5 km and, on clear, cold nights, up to 3.5 km (Bondrup-Nielsen 1984). This allows an observer to locate owls without excessive intrusion. Owl singing locations indicate habitat use and represent potential breeding sites (Meehan 1980, Bondrup-Nielsen 1984, Hayward et al. 1993). Breeding sites vary by vegetation type but boreal owls are obligate cavity nesters (Mikkola 1983).

In this study, we sought to describe habitat used by boreal owls during the 1992–93 mating seasons. We compared owl singing locations with random habitat locations within the study area. The research also illustrates the distribution of boreal owl singing locations in a mosaic of forest patches comprised mainly of lodgepole pine (*Pinus contorta,*

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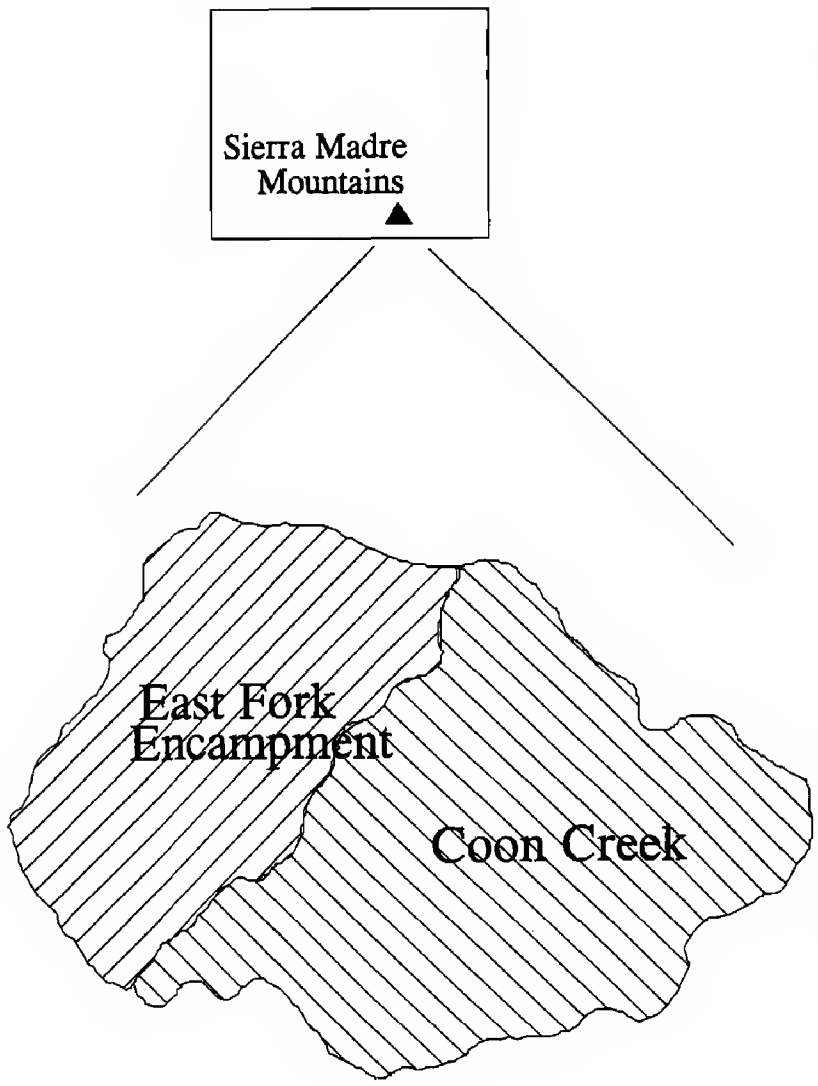


Figure 1. The two watersheds (2526 ha) in the study area located in the Sierra Madre mountains of Wyoming.

LP) and spruce/fir (SF) with interspersed clearcuts. Together with prey and nest hole availability, breeding habitat may be the most critical factor affecting the persistence of owl populations. We specifically describe (1) forest structure at owl singing locations relative to the forested available habitat on the study area, and (2) locations of singing owls relative to clearcuts.

## STUDY AREA

The Blackhall Mountain Study Area is located in the Sierra Madre mountains of southcentral Wyoming in the Medicine Bow National Forest (41°N, 107°W). Two contiguous watersheds, Coon Creek (1615 ha) and the Upper East Fork of the Encampment River (911 ha), are contained in the study area and lie near the Colorado border, 38 km south of Riverside, Wyoming (Fig. 1). Both drainages are heavily-forested mosaics of SF and LP patches, with a few small meadows along the creeks and ridgetops. Lodgepole pine covers 58% of Coon Creek, and 67% of the East Fork for 61% overall. The remaining forests are SF. Half of all stands in both watersheds were classified by the U.S. Forest Service (USFS) as older forests [scoring >38 on Marquardt's (1984) Old Growth Scorecard]. Timber on the Coon Creek watershed was harvested with numerous small clearcuts as part of a water augmentation experiment in the early 1990s. The East Fork was left as an undisturbed control.

Elevation on the study area ranges from 2600–3300 m. Current land uses include logging and grazing; small-scale mining while harvesting for railroad ties occurred historically.

The well-drained soils are 50–150 cm deep. Mean annual precipitation measures 86.4 cm (1983–93), 70% of which falls as snow from late September until late June. Snow survey data from 1993 show a maximum snow depth in late March 1993 of 235.2 cm (Gonyer 1994). The mean annual temperature is 0.6°C, with a low of −42.8°C and high of 30.6°C (Gonyer, 1994).

## METHODS

**Boreal Owl Surveys.** We established 24 transects throughout the study area to locate male boreal owls during the mating season. When possible, we surveyed by snowmobile on unplowed roads and on snowmobile routes traveled by other researchers. We surveyed more remote areas by snowshoe on routes following stream courses and ridges. A total of 122 listening stations (61 on each watershed) on the 24 transects allowed us to detect singing males in all areas of the two watersheds. Although audibility of a boreal owl song is 100% within 700 m (Holmberg 1979), the stations in this study were a maximum distance of 500 m apart to increase detection of singing owls over broken topography.

We surveyed for singing owls for approximately 6 hr after dusk from late February through May of 1992–93. To account for temporal and seasonal variations in singing activity, we surveyed each station at least three times over the two mating seasons in different phases of mating season (early season–late February, mid season–March and April, late season–May), at different times of the night (early, mid, late), during different moon phases (¼, ½, ¾, full), and under different cloud cover (clear, partly cloudy, overcast), snowfall (none, light, moderate), and wind (none, light, medium) conditions. We did not survey under conditions of strong winds and heavy snowfall which were the two factors most affecting calling activity in northcentral Colorado (Palmer 1986).

Because of the potential bias of luring an owl from a singing location toward the listening station, tape play back was not used. All owl locations used in this study represent habitat used by spontaneously singing boreal owls. When owl singing was heard, we either moved toward it until the actual singing tree was located or we identified the location by triangulating the site from two stations (providing the singing continued while we traveled between stations). All singing locations were marked on a 1:24000 U.S. Geological Survey topographic map. We combined all locations from both survey years to describe habitat used by boreal owls during their mating season. Locations that were used more than once during the study were counted as a single location.

**Microhabitat.** The U.S. Department of Agriculture Forest Service Rocky Mountain Forest and Range Experiment Station conducts long-term wildlife research in the study area. A sampling grid of 90 stations in each water-

shed along north-south transects 400 m apart has been established for these wildlife studies. Stations are located 200 m apart, numbered and flagged. We used a subset of those 180 stations to describe available habitat. To achieve a representative subsample, we listed stations from both watersheds by cover type, discarded stations that occurred in clearcuts, and chose every third station on the list until we had at least three times the number of owl singing locations ($N = 22$) and approximately 60% LP stations and 40% SF stations. Each location represented suitable habitat for boreal owls during their mating season.

We determined habitat associations of areas used by boreal owls during their mating season through bird-centered ("singing tree") habitat sampling. We used the James and Shugart (1970) method as modified by Noon (1980) to measure habitat variables at owl singing locations and at available habitat sampling stations. The method employs a 0.04 ha (11.3 m radius) plot. At owl singing locations, the actual "singing tree" was used when it had been found. Otherwise, the closest tree >23 cm diameter breast height (dbh) within 10 m to the triangulated locations was used to represent the "singing tree" and therefore the potential nest tree. The minimum nest tree size used for breeding in Idaho was 23 cm (Hayward et al. 1993).

Within each plot, we classified each tree by species and diameter size class (dbh). We classified each snag >6 m tall into a diameter size class and height, and each downed log >2 m long was placed into a diameter size class and its length measured. We used a 1-factor metric Reloskop to measure basal area of the three tree species ($m^2$/ha) from the center tree and to measure maximum canopy height in the plot. Percent overstory canopy cover was estimated from the average of four measurements in each cardinal direction at the 11.3 m plot edge using a spherical densiometer (Model C). Ground cover was not measured, as the study area was generally dominated by low-growing grouse whortleberry (*Vaccinium scoparium*) and forbs. Aspect was estimated for all plots using the GRASS Geographical Information System (GIS).

**Macrohabitat.** We derived stand size information from USFS maps made prior to clearcutting. Stands were delineated based on vegetative, topographic, and edaphic features. The size of each of the 401 stands within the two watersheds was estimated using a GIS. Stands with owl singing locations were identified by location and site number.

**Distance to Clearcuts.** Because there were no clearcuts in the East Fork watershed, we used the Coon Creek watershed singing locations and available habitat sampling stations ($N = 16$ and 29, respectively) to describe the distance to a clearcut. We measured the distance with a metric tape for plots within sight of a clearcut which was a maximum of 51 m. Plots without a clearcut in sight were categorized as >51 m from a clearcut.

**Data Analysis.** To test if boreal owls used habitats in a nonrandom manner, comparisons of 23 vegetation variables were made at the microhabitat scale. Nonnormal distributions and unequal variances in the data led to the use of nonparametric statistical tests for univariate analysis. Because we used multiple, simultaneous Mann-Whitney $U$ tests, we used Bonferroni-adjusted probability level for $\alpha \geq 0.05$ to $P < 0.0022$.

To examine multivariate patterns in the habitat data, we used an exploratory discriminant function analysis. Eleven variables (basal area, LP >38 cm dbh, ES 15–38 cm dbh, ES >38 cm dbh, SA 0–15 dbh, snags >39 cm dbh, snag height, logs 10–32 cm, logs >32 cm, canopy cover, and canopy height) with low Pearson correlations ($r < 0.55$) were used in three direct discriminant analyses (SPSS, Inc. 1990). Three analyses were done because the two classification groups to separate in the discrimination (owl locations and available sites) were of unequal sample sizes ($N = 22$ and 68, respectively). Three subsamples (S1, S2, and S3) of the available sites were drawn to better balance sample sizes (Williams and Titus 1988). Therefore, each analysis was between all owl locations and one of the three random subsamples (S1, S2, S3) of available sites. Prior probabilities for classification were set for the fraction of cases (habitat plots) in each group Variables with significant structure coefficients (>0.30) from the three subsamples are reported and biologically interpreted (Williams and Titus 1988).

Forest cover type at owl singing locations was determined through cluster analysis (SPSS, Inc. 1990). We used basal area of LP, basal area of SF, the number/ha of LP in two size classes (15–38 cm dbh and >38 cm dbh), and the number/ha of SF in the same two size classes.

Chi-square analyses were done on three categorical variables: aspect, the distance to a clearcut, and cover type. Aspect, and the two categories of distance to a clearcut (<51 m, >51 m), were tested with the chi-square homogeneity test for differences between the expected and observed frequency of use (Jelinski 1991). A chi-square goodness-of-fit test was used to test if the owls used cover type (as determined by cluster analysis) in different proportions than expected based on available habitat proportions of 60% LP and 40% SF (Jelinski 1991, Neu et al. 1974).

The distance to clearcut tests used only the owl singing locations and systematic habitat sampling points in the Coon Creek watershed with the clearcuts ($N = 16$ and 29, respectively). Distance to a clearcut data analysis had two parts. After the chi-square test, a $t$-test was used to test for differences in the measured distances (<51 m) to a clearcut.

We compared the central tendency of stand sizes used by boreal owls to sizes of all stands at the study area. We used a $t$-test to compare the sample stands (owl-use stands) to the population (all stands within the two watersheds).

## RESULTS

Twenty-two boreal owl singing locations were found during the two years of surveys. Six were in undisturbed parts of the East Fork watershed and 16 were in areas of the Coon Creek watershed where there were several small clearcuts. Eight locations were actual "singing trees" where owls were seen in the trees. The remaining 14 locations

Table 1. Comparison of habitat variables at boreal owl singing sites with habitat variables at three nonsinging sites (S1, S2, S3) in the study area.

| VARIABLE | S1 | S2 | S3 | OWL SITE |
|---|---|---|---|---|
| Structure Coefficient | | | | |
| Down logs >32 cm dbh | 0.552 | 0.612 | 0.581 | |
| Canopy height | 0.359 | 0.457 | 0.362 | |
| Snag height | 0.337 | 0.373 | 0.430 | |
| ES >38 cm dbh | | 0.336 | 0.323 | |
| SA <15 cm dbh | 0.369 | | | |
| Means | | | | |
| Down logs >32 cm diam | 3.6 | 7.9 | 3.1 | 7.7 |
| Canopy height | 22.9 | 21.9 | 22.5 | 26.7 |
| Snag height | 10.7 | 9.0 | 8.2 | 16.6 |
| ES >38 cm dbh | 37.2 | 20.0 | 20.7 | 64.7 |
| SA <15 cm dbh | 441.5 | 481.5 | 462.7 | 869.2 |
| Canonical correlation | .777 | .816 | .805 | |
| % Correctly classified | 92.98 | 96.49 | 94.74 | |
| Centroids | 1.53 | 1.75 | 1.68 | |
| | −.961 | 1.10 | 1.05 | |

were found after hearing owls at two listening stations. All 22 locations were treated equally.

**Microhabitat.** Four of the 23 variables tested in univariate analysis were significantly different between boreal owl locations and available habitat. The four variables were: basal area of ES (12.4 $m^2$/ha vs 6.6 $m^2$/ha, owl locations vs available habitat, respectively, $P < 0.0001$); basal area of SA (12.4 $m^2$/ha vs 8.7 $m^2$/ha, $P = 0.0005$); canopy height (27 m vs 23 m, $P = 0.0004$); and large, downed logs (7.7 vs 3.7, $P < 0.0001$).

Taller overstory canopy, tall snags, and many large (>32 cm dbh) downed logs were identified as important forest characteristics in the discriminant analyses (Table 1). The three analyses (S1, S2, S3), based on 11 forest structure variables, accounted for 60%, 67%, and 65% of the variance within the groups, respectively. The single canonical function generated for each analysis significantly distinguished between available habitat and sites used by singing boreal owls ($P < 0.001$). The first subsample (S1) added the number/ha of small SA trees to the function while S2 and S3 added the number/ha of large (>38 cm dbh) ES trees to the function. Discriminant function analyses provided 93%, 96.5%, and 95% correct classification, respectively, suggesting substantial differences between available habitat and owl singing sites.

Snag height (somewhat correlated with large diameter snags, $r = 0.45$) was also identified in the discriminant analysis. At owl singing locations, large diameter snags (>39 cm dbh) were generally taller (21.23 m, range 12–31.8 m) than medium (18–39 cm dbh) diameter snags (15.6 m, range 6–26.8 m, $P < 0.013$). Large diameter snag density at the Blackhall Mountain study area was estimated at 19 ± 21.7 snags/ha at boreal owl singing locations, while the available habitat had an estimated 8 ± 16 snags/ha.

Cluster analysis of basal area and tree density data for owl location plots determined the forest cover type. Five locations were in the LP cover type and 17 in the SF cover type.

The SF cover type was used by boreal owls more frequently than its proportional availability (40%) at the study area ($\chi^2 = 13.54$, df = 1, $P < 0.005$). The lodgepole cover type was used less than its proportion of availability (60%).

The aspect at boreal owl singing locations did not differ from aspects at sampled available habitat locations ($\chi^2 = 9.19$, df = 7, $P = 0.239$). The chi-square test of homogeneity compared observed and expected frequencies between owl locations and available sites ($N = 22$ and 68, respectively).

**Macrohabitat.** The 22 owl singing locations occurred in 16 stands. One large (122.8 ha) SF stand had five locations, two LP stands had two singing locations each, and the remaining 13 stands had one singing location per stand. It is quite possible that the same bird used several of these locations.

Boreal owl breeding season home ranges have been reported to vary between 240–352 ha for two birds in Colorado (Palmer 1986) and to average 1451 ± 552 ha in Idaho (Hayward et al. 1993).

At this scale, boreal owl singing locations were almost equally distributed by cover type into SF stands (55%) and LP stands (45%). This differed from plot level cover type designation (from cluster analysis) for three of the 16 stands used by singing boreal owls.

The mean stand size at the study area (6.9 ± 13.7 ha) differed from mean stand size at owl locations (35.5 ± 36.4 ha, $P = 0.0067$). Stands in the available habitat ranged in size from 0.005–122.8 ha; stands used for singing ranged from 0.17–122.8 ha.

**Distance to Clearcuts.** The 227 small clearcuts in the Coon Creek watershed have left 82% of the forest within 200 m of a clearcut. The maximum possible distance from a clearcut is 270 m (E. O'Doherty pers. comm.). Boreal owl singing locations did not differ in the distance to a clearcut from the random available habitat locations ($\chi^2 = 1.65$, df = 1, $P = 0.199$). The chi-square test of homogeneity tested the two categories of distance to a clearcut (<51 m, >51 m) for boreal owl singing locations ($N = 16$ in this watershed) and random available habitat sampling points ($N = 29$ in this watershed). Further, a *t*-test between the measured distances (<51 m) showed no difference ($P = 0.65$) in the mean distance to a clearcut between owl singing locations (29 ± 12.8 m) and the available habitat sampling points (31.7 ± 12.9 m) in the Coon Creek watershed. The closest "singing tree" to a clearcut was 15 m and the average distance was 27 ± 30 m.

## DISCUSSION

The majority of owl singing locations at the Blackhall Mountain study area were in the SF cover type with microhabitat structure typical of mature or old-growth forests (large downed logs, a high overstory canopy, tall snags, large ES trees, and small fir trees). Large snag (>39 cm dbh) density at owl singing locations, greater than in either the available habitat or other study areas in Colorado (Palmer 1986, Ryder et al. 1987) or Idaho (Hayward et al. 1993), increases the potential for suitable nest sites for boreal owls. Dense ES forests also offer protection from predators such as pine marten (*Martes martes*) (Korpimäki 1988) and larger birds of prey (Mikkola 1983). The singing locations in LP cover types had large lodgepole trees instead of ES trees, as did boreal owl singing locations in LP stands on the Beaverhead National Forest in Idaho (Hayward et al. 1993). Hence, some LP stands appear to have adequate forest structure for boreal owls to use as singing sites.

Canopy height and large downed logs were identified as important differences between owl singing sites and random sites in both multivariate and univariate tests. Large downed logs are an important component of old-growth forests (Maser et al. 1979, Meslow et al. 1982) and were correlated with boreal owl mating habitat on our study area. Webb (1982) also described three of his five boreal owl locations in northern Colorado as having "much fallen timber." Although studies on boreal owl foraging sites did not report the log component directly, the highest numbers of foraging sites were in mature or older SF forests that support the owl's primary prey, the red-backed vole (Palmer 1986, Hayward et al. 1993). The old-growth locations used by boreal owls during the mating season at the Blackhall Mountain study area may also be used for foraging.

Microhabitat at owl singing locations indicated that SF cover types were used most often by boreal owls. Similarly, Palmer (1986) found a higher density of boreal owls in Colorado's high elevation SF though four other habitat types were available. Only in years with an abundance of boreal owls at his study area were lower elevation mixed forest habitat types used. He suggested that the SF is optimum habitat in the Cameron Pass area (Palmer 1986) 87 km south of the study area.

Hayward et al. (1993) found from surveys throughout the northern Rockies that a majority of boreal owl locations were in SF habitat types. In Idaho, the owls bred more often in mixed conifer and aspen habitats and, in the wilderness study site, did not nest in boxes hung in LP stands. At Blackhall Mountain, the aspen cover type was not available, and some LP stands were used for singing. Like some of the boreal owl singing sites in Colorado (Palmer 1986, 1987), sites used in Blackhall Mountain during mating season may be suboptimal. Without knowing the density and reproductive success of boreal owls at the Blackhall Mountain study area and in the surrounding habitat, inferences about the use of areas of lesser quality could not be made. Hayward et al. (1993) suggested that suboptimal habitat may still be important at a regional and metapopulation scale.

The discrepancy in cover-type designation between USFS maps and the cluster analysis for three of the 16 stands used by singing boreal owls is probably due to the difference in scale. This situation illustrates the importance of collecting habitat data on the ground rather than from a large-scale map.

The wide range of stand sizes used by singing boreal owls at Coon Creek suggests that even small stands provide forest structure to allow the owls to attempt breeding. However, adjacent stand conditions (clearcut, cover type, stand size, etc.) were not investigated though they may have had a significant influence. Also, the present and future degree of suitability of these small sites was unknown. Rosenberg and Raphael (1986) found that small patches of old-growth support forest-interior species, although their smallest stand measured 5 ha. In the long term, as the amount of habitat available is reduced, owl populations will decline (Hayward et al. 1993).

The choice of habitat made by boreal owls may be of ecological importance to their survival. Microhabitat selection may help avoid predation by larger owls. Food supply may be easier to obtain thereby conserving energy in this very harsh environment.

Boreal owls in the Coon Creek watershed sang in trees near clearcut edges. The "singing tree" and microhabitat data suggested that old-growth conditions are found near the edges of clearcuts. Owls may use edges for several other reasons. If the location was used for nesting prior to the clearcuts, it may continue to be used because of nest site tenacity by male boreal owls when suitable nest sites are scarce as in Sweden (Lundberg 1979) and western Finland (Korpimäki 1988). If clearcutting eliminated more suitable sites and packed owls into remaining forest, suboptimal locations may be used. Korpimäki (1987), for example, found that male boreal owls used nest holes just over 1 km from the edge. We speculate that stand structure is more important than any factor relating to edge. Boreal owl populations are associated with old growth characteristics including large, tall trees. Large downed logs are an indirect indicator of such stands and may also provide foraging sites for boreal owls.

Perhaps the most intriguing finding of this study is that the majority of the singing locations were in the watershed containing clearcuts. Korpimäki (1988) found that Tengmalm's owls in western Finland preferred voles (*Microtus* sp.) that occupied clearcut areas where snow melted earlier than in woodlands. During breeding, which starts as the snow is melting, these owls mainly hunted in fields. He found that agricultural lands interspersed with productive ES forests lowered the variability of food supply which benefitted breeding owls. The mosaic of openings and forests provided several species of small mammals and therefore balanced the seasonal and year-to-year population fluctuations. Foraging boreal owls in Norway avoided clear-fellings that had higher prey densities than old-forest stands because the higher, denser vegetation made prey less accessible (Sonerud et al. 1986). Recent clearcuts in the Coon Creek watershed may provide greater prey availability for boreal owls at forest edges than in the undisturbed East Fork watershed. It is possible the owls have benefitted from some clearcutting in the short-term.

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# NEST-SITE AND AERIAL MEETING POINT SELECTION BY COMMON BUZZARDS (*BUTEO BUTEO*) IN CENTRAL ITALY

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ABSTRACT.—A nesting population of common buzzards (*Buteo buteo*) was studied in a mountainous area of central Italy from 1988–92. Nesting density averaged 19.74 pairs/100 km$^2$, and the average minimum distance between pairs was 1.4 km (SD = 0.432). Mean number of young fledged/successful nest was 1.78 for all years combined. Of nests examined, a significantly ($P = 0.001$) larger proportion were on slopes facing to the northeast (73.3%), most were on the mid-portions of slopes (60%), and were built at the intersections between tree branches and tree trunks (86.6%). Other factors including elevation, the angle between tree trunks and branches, tree height, tree crown volume, the distance of nests from a forest edge, the distance of the nest from areas of timber harvesting, and the average trunk spacing were also important variables in terms of nest placement. The distance of aerial meeting sites (areas where a group of at least three buzzards regularly soared, tumbled together, and chased each other) from neighboring nest sites and maximum slope were also important factors in the choice of these gathering points.

KEY WORDS: *nest-site selection; buzzard;* Buteo buteo; *reproduction; aerial meeting points; central Italy.*

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Sitios de nidificación y selección de puntos de reunión aérea por *Buteo buteo* en Italia central

RESUMEN.—Una población nidificante de *Buteo buteo,* fue estudiada en un área montañosa de Italia central, desde 1988 a 1992. La densidad promedio de nidificación fue de 19.74 parejas/100 km$^2$, la distancia mínima promedio entre parejas fue de 1.4 km (DS = 0.432). El número medio de juveniles volantones/nido exitoso fue 1.78 para todos los años combinados. De los nidos examinados, una significativa proporción (P = 0.001) estaba sobre laderas de exposición noreste (73.3%), la mayoría estaba sobre la porción media de las laderas (60%) y fueron construidos en la intersección de ramas y troncos de árboles (86.6%). Otros factores que incluyeron elevación, ángulo entre ramas y troncos, altura del árbol, volumen de cosecha arbórea, distancia de los nidos al borde del bosque, distancia del nido a áreas de cosecha y el espacio promedio entre troncos, fueron importantes variables respecto a la ubicación del nido. La distancia de sitios aéreos de reunión (áreas donde un grupo de al menos tres individuos regularmente remontaban el vuelo, caían juntos y se perseguían unos a otros) a sitios de nidificación vecinos y máxima inclinación también eran factores importantes en la elección de estos puntos de reunión.

[Traducción de Ivan Lazo]

Studies on habitat use by birds show that they nest in those portions of the available natural environment which best suit their primary living requirements (Hilden 1965, Morse 1980, Cody 1985). Common buzzards (*Buteo buteo*) have been the focus of numerous and diversified studies, conducted in most of their range (Mebs 1964, Tubbs 1974, Rockenbauch 1975, Weir & Picozzi 1975, 1983, Picozzi & Weir 1976, Arce Velasco 1987); however, few data are available on their selection of nesting habitat (Kostrzewa 1987, Jedrzejewski et al. 1988, Kostrzewa & Kostrzewa 1988, Hubert 1993). This study was designed to characterize breeding density, reproductive success, and nest-site selection in a common buzzard population in a mountainous area. In addition, we sought to provide data on the selection and use of aerial meeting sites of buzzards (Tubbs 1974).

## STUDY AREA

The study was conducted in a mountainous area measuring 400 km$^2$ between the Latium and Abruzzo regions

of central Italy. Elevation of the area ranged from 508–1820 m. The landscape consisted of a mosaic of habitat types including forests, pastures, clearings, and piedmont crop areas. Forested areas were the most common cover type covering approximately 35.5% of the total area (I.S.T.A.T. 1991). Dominant tree species were *Castanea sativa, Quercus cerris, Q. pubescens, Pinus nigra,* and *Fagus sylvatica.* Most of the forested area was being used as coppice.

## METHODS

We mapped forested areas using 1:25 000 scale maps and 1:10 000 scale aerial photos. Because common buzzards are found in a variety of habitats (Tubbs 1974, Cramp and Simmons 1980), all forested areas were surveyed for breeding pairs. We located occupied nesting areas by observing territorial flights, nuptial displays, nest building during the early stages of the breeding period (February–March), and prey deliveries to nests during the nestling period (June). We also used recorded playbacks of common buzzard calls during March, April, June, and October (Cerasoli and Penteriani 1992) to locate occupied nesting areas.

To assess reproductive success, we observed occupied nests from fledging until the young left the nest area (buzzard fledging period: 48–62 d, Cramp and Simmons 1980), and production was calculated as the mean number of fledgings/successful nest. To estimate nesting density, we used nearest neighbor distance (Newton et al. 1977).

Nest-site characteristics were analyzed on two levels. Level 1 analysis assessed features of nest trees and the nests themselves and Level 2 assessed habitat features surrounding the nest area (Table 1). Level 1 features were measured using a tree caliper, metric tape and compass.

Level 2 analysis used circular, nest-site plots with 30 m radii centered on nest trees (James and Shugart 1970, Reynolds et al. 1982, Titus & Mosher 1987, Jedrzejewski et al. 1988). Features of trees in plots were sampled using four, 30 m transects radiating from the nest tree at right angles to each other and following the four cardinal compass directions. Trees intercepted by the lines were measured using the line intercept method (Mueller-Dombois & Ellenberg 1974, Burnham et al. 1980, Bonham 1989). To identify possible habitat selection, we used a point-centered-quarter method (Mueller-Dombois & Ellenberg 1974, Bonham 1989) consisting of four plots established in each of the cardinal compass directions, 60 m from nest trees. These four plots were 60 m in diameter and four, 30 m transects radiated from the center of each plot in each of the cardinal compass directions. Canopy cover was measured along the four transect lines in each plot by estimating percentage of sky not obstructed by vegetation in black & white photos taken with a camera placed horizontally on a tripod and fitted with a 28 mm, f.3.5 lens. Nest-site characteristics were measured at a total of 15 occupied nests for Level 1 analysis, and at 13 occupied nests for Level 2 analysis.

We also measured habitat characteristics within a 0.5 km radius of eight aerial "rendezvous" sites (areas where a group of at least three common buzzards were regularly seen soaring, tumbling together and chasing each other, Tubbs 1974) to determine if the selection of these meeting sites was dependent on neighboring nest-site location and/or topographic features facilitating flight and minimizing energy requirements (Cody 1985). In this case, we used the point-centered-quarter method with four, 1-km diameter sample plots tangent to the rendezvous site and centered on the cardinal compass directions Percentage slope was calculated inside the plots and along slopes using the number of contour lines on topographic maps of the area. Using this method, maximum percentage slopes had the greatest number of contour lines and minimum percentage slopes had the fewest contour lines. By definition, rendezvous sites had to contain at least three common buzzards. The number of additional common buzzards at a rendezvous site was treated as the dependent variable in a multiple regression model. Independent variables were: (1) distance of the plot center from the nearest nest and (2) percent slope at the center of the plot.

Data were not in consistent units of measurement so we converted them to nondimensional index numbers. Qualitative variables, such as tree species and slope exposure were also transformed into indexes. We used (1) principal component analysis (PCA) to scale down the number of variables; (2) cluster analysis and analysis of variance (ANOVA) to test for nest-habitat selection; (3) chi-square tests to examine the distribution of nests relative to slope position and exposure; (4) chi-square and Mann-Whitney tests to compare characteristics of common buzzard nest sites and sample plots, and rendezvous sites and sample plots; and (5) multiple linear regression for characterization of rendezvous sites (Sokal and Rohlf 1981).

## RESULTS

We found 15 pairs of breeding common buzzards in the 91.18 km$^2$ study area, for a density of 19.74 pairs/100 km$^2$. Minimum distance between the pairs averaged 1.4 km (SD = 0.43, range = 0.85–1.82). Egg-laying took place during the second week of April and fledging occurred in the first half of June. In only one case were eggs laid during the third week of April. Annual productivity of breeding pairs was 1.78 fledgings/successful pair (SD = 0.16, range = 1.62–2.00).

Common buzzards nested in a diversity of trees. Of 15 occupied nests, five (33%) were in *Castanea sativa* trees, three (20%) in *P. nigra* trees, two (13%) in *Q. cerris* trees, and one each (6.7%) was in a *Picea excelsa, Ostrya carpinifolia, F. sylvatica, Q. pubescens,* and *Populus* spp. tree. Eleven (73.3%) of the nest trees were on slopes that faced northeast and they were on the mid-portions of slopes. Thirteen nests were situated at the intersection between a tree branch and the trunk, and the remaining two nests were on lateral branches. Seven of the 24 variables measured at nests were significantly different from the same variables at measured sample plots: elevation ($F = 2.82$; $P =$

Table 1. Sample means and standard deviations of characteristics of nest-site and sample plots for common buzzards in central Italy.

| | NEST SITES (RANGE) | CONTROL PLOTS (RANGE) | TEST STATISTIC |
|---|---|---|---|
| Level 1 Analysis ($N$ = 15) | | | |
| Tree DBH (cm) | 27.77 ± 7.27 (18–42) | — | $\chi^2$ = 22.82 |
| Tree height (m) | 17.58 ± 2.96 (14–25) | — | $\chi^2$ = 21.35 |
| Nest height (m) | 12.7 ± 2.77 (8.5–15.5) | — | $\chi^2$ = 7.26 |
| Relative height of nest in tree (%) | 72.77 ± 1 (52.5–91.4) | — | $\chi^2$ = 30.32* |
| Relative height of nest in crown (%) | 48.85 ± 25.58 (5.88–92) | — | $\chi^2$ = 160.96*** |
| Number of branches supporting nest | 3.92 ± 1.5 (2–7) | — | $\chi^2$ = 3.03 |
| Distance to nearest timber harvest (m) | 40.28 ± 23.08 (4–71) | 5.74 ± 21.76 (0–141) | $U$ = 240, $z$ = −1.4 |
| Distance to nearest forest trail (m) | 28.23 ± 19.4 (2–74) | 42.5 ± 26.46 (0–98) | $U$ = 433, $z$ = −0.93 |
| Distance to nearest water (m) | 93.8 ± 59.77 (42–203) | 102 ± 51.39 (6–250) | $U$ = 302, $z$ = −0.15 |
| Distance to nearest woodland edge (m) | 67.59 ± 48.41 (4–120) | 72.4 ± 24.75 (0–182) | $U$ = 273, $z$ = −0.61 |
| Level 2 Analysis ($N$ = 13) | | | |
| Elevation (m) | 927.33 ± 122.88 (770–1230) | 989.3 ± 142.64 (750–1250) | $U$ = 333, $z$ = −0.01 |
| Tree dbh (cm) | 11.94 ± 10.42 (2–33) | 8.92 ± 6.76 (2–90) | $U$ = 340,** $z$ = −3.36 |
| Tree height (m) | 10.74 ± 3.04 (3.5–25) | 7.58 ± 3.56 (3.1–18) | $U$ = 211,** $z$ = −3.37 |
| Height of trunk without branches (m) | 5.15 ± 2.25 (1.1–11) | 2.52 ± 1.47 (1.1–8.63) | $U$ = 163, $z$ = −1.31 |
| Number branches in tree | 21.55 ± 9.02 (14–55) | 10.8 ± 9.69 (8–55) | $U$ = 511,* $z$ = −3.26 |
| Angle between trunk and branches (°) | 64.08 ± 7.74 (50–90) | 37.6 ± 16.97 (30–90) | $U$ = 169, $z$ = −1.63 |
| Tree crown volume ($m^3$) | 170.02 ± 68.51 (29.44–463) | 42.59 ± 37.26 (1.07–278.56) | $U$ = 46,*** $z$ = −4.04 |
| Trunk spacing (m) | 2.38 ± 0.85 (0.88–3.53) | 1.75 ± 0.86 (0.84–3.29) | $U$ = 107,* $z$ = −2.58 |
| Canopy cover (%) | 16.07 ± 9.42 (2–50.8) | 76.12 ± 28.31 (5–100) | $U$ = 139,* $z$ = −2.71 |

* $P < 0.01$, ** $P < 0.001$, *** $P < 0.005$.

0.046), angle between trunk and branches (F = 73.28; $P$ = 0.0001), nest-tree height (F = 98.24; $P$ = 0.0001), tree crown volume (F = 87.16; $P$ = 0.0001), distance of nest tree from forest edge (F = 6.06; $P$ = 0.001), distance of nest tree from timber harvesting (F = 13.84; $P$ = 0.0001), and average trunk spacing (F = 44.62, $P$ = 0.0001). Single linkage analysis (Sneath and Sokal 1973) did not form separate groups of nest trees on the basis of these seven variables but Ward's analysis (Everitt 1974) identified four groups of nest-site plots which enabled us to identify each nest-site variable as belonging to a group with a unique pattern of variables. Groups 1 and 3 contained 25 and five

Table 2. Average (±SD) of the seven main components (PCA) in the four groups of nest-site plots identified by the Ward's method.

| VARIABLE | GROUP 1 | GROUP 2 | GROUP 3 | GROUP 4 |
|---|---|---|---|---|
| Elevation (m) | 921.6 ± 121.5 | 993.1 ± 30.4 | 1057 ± 163.8 | 897.5 ± 113.74 |
| Angle between trunk and nest branch (°) | 13.8 ± 20.5 | 69.8 ± 10.3 | 6 ± 13.4 | 58.3 ± 5.2 |
| Tree height (m) | 1.9 ± 2.9 | 13.3 ± 3.2 | 0.7 ± 1.6 | 18.9 ± 3.5 |
| Tree crown volume ($m^3$) | 1.0 ± 2.0 | 56.5 ± 41.3 | 0.2 ± 0.5 | 283.6 ± 95.8 |
| Nest distance from forest edge (m) | 38.8 ± 25.4 | 61.9 ± 42.5 | 106 ± 24.1 | 73.3 ± 54.4 |
| Nest distance from timber harvesting (m) | 29.8 ± 18.6 | 43.1 ± 23.9 | 98 ± 24.9 | 37.5 ± 22.3 |
| Trunk spacing (m) | 0.4 ± 0.5 | 2.1 ± 0.8 | 0.2 ± 0.4 | 2.5 ± 0.7 |

plots, respectively, and none had nest sites. Groups 2 and 4 contained 29 and six plots, respectively, and had nine (31%) and four (66.7%) nests.

For each of the seven main components, the average in each group was determined (Table 2). Averages for groups 2 and 4 that contained nest plots were 993.1 m and 897.5 m for elevation, 69.83° and 58.33° for the angle between the trunk and the branch supporting the nest, 13.26 m and 18.91 m for nest-tree height, 56.47 $m^3$ and 283.58 $m^3$ for the tree crown volume, 61.86 m and 73.3 m for the distance of the nest tree from the nearest forest edge, 43.06 m and 37.5 m for the distance of the nest tree from the nearest timber harvesting; and 2.08 m and 2.47 m for trunk spacing.

Mean values for several variables were higher in nest-site plots than in sample plots. There was a significant difference for tree height ($U = 211$, $z = -3.37$, $P = 0.0008$), tree crown volume ($U = 46$, $z = -4.04$, $P = 0.0002$), trunk spacing ($U = 107$, $z = -2.58$, $P = 0.01$), nest-tree diameter ($U = 340$, $z = -3.36$, $P = 0.0009$), number of branches in the nest tree ($U = 511$, $z = -3.26$, $P = 0.002$), and canopy cover ($U = 139$, $z = -2.71$, $P = 0.008$) between nest-site plots and sample plots (Table 1). We also found statistically significant differences between tree diameter ($U = 352$, $z = -5.24$, $P = 0.001$) and tree height ($U = 257$, $z = -47.38$, $P = 0.001$) for nest trees and other trees inside the nest plot.

Table 3. Means (±SD) of characteristics of eight "rendezvous" sites and sample plots.

| | RENDEZVOUS SITES (RANGE) | SAMPLE PLOTS (RANGE) | TEST STATISTIC |
|---|---|---|---|
| Distance from nests (m) | 770.8 ± 496.8 (350–1675) | 1187 ± 469 (575–2275) | $U = 27$, $z = -1.91$ |
| Maximum % slope | 49.9 ± 12.3 (40–75) | 52.2 ± 18 (2.5–85.7) | $U = 119$, $z = -0.34$ |
| Minimum % slope | 20.1 ± 6 (10.3–28.6) | 19.1 ± 9.7 (7.14–41.7) | $U = 99$, $z = -1.06$ |

Rendezvous points of common buzzards averaged 770.8 m (SD = 496) from neighboring nest sites (Table 3). Regression coefficients of independent variables derived from the multiple linear regression model were negative in terms of distance of rendezvous site plots from neighboring nest sites ($r = -0.0003$, $P = 0.01$; $\bar{x} = 770.8$ m, SD = 496.84) and for minimum slope ($r = -0.002$, $P = 0.13$; $\bar{x} = 20.1$ m, SD = 6.16), and positive for maximum slope ($r = 0.002$, $P = 0.14$; $\bar{x} = 49.9$ m, SD = 12.28). The highest correlation was obtained for the distance between the rendezvous point and the nearest neighboring nest site. These data showed that the distance of rendezvous sites from neighboring nest sites was the most significant factor in the choice of these gathering points. Maximum slope may have also affected site selection.

## DISCUSSION

We found that common buzzards showed a distinct tendency to select nest trees located in the mid-portion of northeastern-facing mountain slopes. They built their nests at the intersection between a tree branch and the trunk, approximately ⅔ the way up the tree. This tendency was also ob-

served by Tubbs (1974), Rockenbauch (1975), A.C.I.N.E.R. (1979), and Hubert (1993). Easy access to nests appears to be a key factor in nest placement. Nest placement between tree branches and trunks facilitates frequent trips made by adults to and from nests with food, as well as early flights of recently fledged young (Tubbs 1974, Hubert 1993). Other factors influencing nest-site selection are the presence of large branches and abundant foliage, both of which protect the nest from predators and weather (Tubbs 1974).

The tendency to use northern slopes has also been noted by Manzi & Pellegrini (1989). These slopes may provide cooler temperatures and less sunlight in the nest themselves, and the denser tree cover on northern slopes may increase protection for nests. Placement of nests midway up northern slopes, in the tallest trees available, may also increase the accessibility of nests to both adults and fledglings saving energy and reducing food demands (Weir & Picozzi 1975). Elevated nests may also provide vantage points from which hunting areas can be more easily watched (Tubbs 1974).

Our analysis indicated there were six characteristics which best described selection of nest trees by common buzzards: the angle between the nest tree branch and trunk, the height of the nest tree, the tree crown volume, the distance of the nest from the nearest forest edge and timber harvesting area, and the average trunk spacing. Selection of taller trees, with denser canopies and larger average trunk spacing has also been noted by Hubert (1993).

The proximity of nests to timber harvesting areas and areas with forest edges suggests that nest tree selection may also be influenced by the availability of nearby foraging areas (Tubbs 1974, Picozzi & Weir 1976, Cramp and Simmons 1980, Jedrzejewski et al. 1988, Hubert 1993) and their accessibility to both adult and immature buzzards (Roche 1977 and Hubert 1993).

Common buzzards apparently use rendezvous points as social gathering areas to designate territorial boundaries of neighboring pairs (Tubbs 1974). Our analysis showed that in selecting these areas, common buzzards chose steep slopes that contribute to the formation of rising air currents and facilitate high-altitude turns at these meeting sites (Weir & Picozzi 1975).

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# ENVIRONMENTAL CONTAMINANT LEVELS IN SHARP-SHINNED HAWKS FROM THE EASTERN UNITED STATES

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ABSTRACT.—We examined contaminant levels in tissue samples of sharp-shinned hawks (*Accipiter striatus*) collected in the eastern U.S. from 1991–93. We report concentrations of aldrin, cis-nonachlor, p,p'-DDE, dieldrin, heptachlor epoxide, mirex, oxychlordane, PCB, aluminum, lead, and mercury detected in 23 blood, 10 brain, and 31 liver samples. DDE, PCB's, and mercury were detected most often and in highest concentrations. No contaminants were present at concentrations that might cause mortality with the possible exception of one individual with high oxychlordane residues in the liver. It is not known, however, at what levels these contaminants might impair reproduction in sharp-shinned hawks. Migration count data (declining sharp-shinned hawk numbers in the East, stable in the Midwest) coupled with contaminant data (higher DDE levels in blood in eastern sharp-shins than in midwestern) do not rule out the possibility that contaminants may be impairing reproduction in the eastern population, although our data suggest that this is unlikely. Further study of contaminant levels in sharp-shinned hawks with concurrent research on their productivity and on prey availability is necessary. This species also may be an important indicator species for monitoring contaminant levels because of their high position in the food chain.

KEY WORDS: Accipiter striatus; *contaminants*; *sharp-shinned hawks*

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Niveles de contaminantes ambientales en *Accipiter striatus* en el este de los Estados Unidos

RESUMEN.—Examinamos niveles de contaminación en muestras de tejido de *Accipiter striatus,* colectados al este de los Estados Unidos durante 1991 y 1993. Reportamos concentraciones de aldrin, "cis-nonachlor," PCB, aluminio, plomo y mercurio, detectadas en 23 muestras de sangre, 10 de tejido cerebral y 31 de hígado. Tanto DDE, PCB y mercurio fueron detectados más a menudo y en altas concentraciones. Sin embargo, no es conocida la concentación en los que estos contaminantes podrían dañar la reproducción en individuos de *A. striatus.* Los datos de conteos migracionales (declinación del número de *A. striatus* en el este y estabilidad en el medio-oeste) acoplados con datos de contaminantes (niveles de DDE sanguíneos mayores en *A. striatus* del este que en el medio-oeste), no descartan la posibilidad de que contaminantes puedan estar dañando la reproducción en poblaciones del Este, aunque nuestros datos sugieren que esto es improbable. Esta especie puede ser una importante indicadora para monitorear niveles de contaminación, considerando su alto nivel trófico en la cadena alimentaria.

[Traducción de Ivan Lazo]

Recently, counts of migrant sharp-shinned hawks (*Accipiter striatus*) at hawk migration watch sites in the eastern U.S. have declined, while counts conducted in the Midwest and West have remained steady or increased (Kerlinger 1992). The coastal watch site at Cape May, New Jersey began reporting declines in sharpshin counts about 1986 (Dodge 1992, Kerlinger 1992, Panko 1992). In contrast, the

mountain ridge site at Hawk Mountain Sanctuary did not note a decrease until 1990 with significant decreases occurring from 1991–93 (Laura 1992, Viverette et al. 1996). There is no indication that changes in timing or numbers of cold fronts were responsible for the declines. Numbers of sharp-shinned hawks at Hawk Mountain during the fall are not correlated with the movement or number of cold fronts (Allen et al. 1996).

Hawks banded at Cape May and Hawk Mountain have an 80% overlap in breeding range and nearly identical wintering ranges in the southeastern U.S. (Struve and Goodrich 1992, Viverette et al. 1996). Thus birds counted at the two sites are predominantly from the same population. Hawk Mountain, however, records over 60% adult sharp-shinned hawks during fall migration (Goodrich, unpubl. data), while Cape May records over 80% juveniles (Clark 1985). If sharp-shinned hawk populations were being adversely affected by poor reproduction, numbers of juveniles would decrease first, followed by declines in the adult population. Based on the closely related Eurasian sparrowhawk (*Accipiter nisus*) (Newton 1986), it is estimated that most sharp-shinned hawks may not breed until their third year (Johnsgard 1990); thus a 3–4 year delay is expected.

A similar pattern in declining counts occurred in bald eagles (*Haliaeetus leucocephalus*) when numbers of juveniles declined several years before counts of adults declined (Bednarz et al. 1990). For eagles, declines resulted because reproduction was impaired by DDT and other environmental contaminants. Perhaps contaminant exposure and impairment of reproduction may explain the reduction in sharp-shinned hawk counts at northeastern coastal watch sites several years before any reduction at other inland ridge sites.

Despite the U.S. ban on DDT in the 1970s, raptors continue to be exposed to persistent organochlorine compounds in both the U.S. and Canada (Court et al. 1990, Peakall et al. 1990, Porter 1993). Sublethal doses of contaminants in birds can and do impair reproduction (Peakall 1970). In New Jersey, Steidl et al. (1991) detected DDT-related eggshell thinning in nesting osprey (*Pandion haliaetus*) during 1985–88 and California's peregrine falcons (*Falco peregrinus*) had high egg levels of DDE throughout the 1980s (Clark 1990).

Raptors are particularly susceptible to toxic chemicals that bioaccumulate through each trophic level. Sharp-shinned hawks in particular feed high on the food chain by preying primarily on small birds (Storer 1966). Thus, they are an important species for monitoring bioaccumulation of contaminants in terrestrial systems (Elliott and Shutt 1993). Contaminant levels in sharp-shinned hawks are not well documented, particularly levels measured during recent years when population declines were observed. Noble and Elliott (1990) reported contaminant levels in eggs of sharp-shinned hawks collected in eastern Canada in 1980–88. Elliott and Shutt (1993) reported contaminant levels for sharp-shinned hawk blood samples collected in 1985–89 from two migration sites (Whitefish Point, Michigan and Hawk Cliff, Ontario) on the eastern Great Lakes. Herein, we present results from a study to examine concentrations of various environmental contaminants in blood, brain, and liver tissues of sharp-shinned hawks from the eastern U.S.

## STUDY AREA AND METHODS

Blood samples were collected from 21 sharp-shinned hawks trapped on the Kittatinny Ridge in eastern Pennsylvania and at Cape May, New Jersey, during southward migration September through November 1991 and 1992. Although trapped with lure traps, they are representative of healthy, migrating sharp-shinned hawks from the eastern U.S. (Powers et al. 1994). Trapping locations are described in Clark (1985). Approximately 1 ml of blood was collected from the jugular or brachial vein with a sterile needle and syringe and immediately transferred to a heparinized vacutainer. Blood samples immediately were placed in a cooler with ice and were transported to the lab each day. Blood samples were centrifuged and the plasma was separated from the remaining blood components with a clean glass pipette. The plasma was placed into a glass vial with a lid made of inert material (teflon or aluminum), then frozen for later analysis. The amount of plasma in each sample varied from 0.01–0.65 g with most samples near 0.5 g.

Carcasses of 31 sharp-shinned hawks (19 adult, 12 juvenile) from the eastern U.S. were obtained throughout the fall and winter of 1992–93. Carcasses of sharp-shinned hawks that were killed during collisions with windows or automobiles were collected from rehabilitation centers. All carcasses were refrigerated and shipped to Hawk Mountain Sanctuary overnight on dry ice. Liver and brain tissues were removed for contaminant analyses All tissue samples were stored in a freezer and shipped frozen to the analytical laboratory on dry ice.

We analyzed 21 blood, 10 brain, and 31 liver samples from the eastern population for various compounds including aldrin, cis-nonachlor, p,p′-DDE, dieldrin, heptachlor epoxide, mirex, oxychlordane, and PCB (Table 1). We also analyzed samples for aluminum, cadmium, chromium, lead, mercury, and selenium. Not all samples were analyzed for each compound or heavy metal. Samples collected in 1992 and 1993 were tested for contaminants at Hazleton Environmental Laboratory in Madi-

Table 1. Number of samples (*N*) and values (ppm wet weight) of pesticide and heavy metal compounds analyzed in sharp-shinned hawk tissues from the eastern U.S., 1991–93. The first line of data for each compound is the number of samples with detectable levels of the contaminant and the values detected. The second line is the number of samples with values below the detection limit and the values of the detection limits.

| | Blood | | Brain | | Liver | |
|---|---|---|---|---|---|---|
| Compounds | *N* | Values | *N* | Values | *N* | Values |
| Aldrin | 0 | | 0 | | 0 | |
| | 13 | <0.06–<4.00 | 8 | <0.05–<0.71 | 9 | <0.03–<0.08 |
| Cis-nonachlor | | | 0 | | 2 | 0.40–1.60 |
| | | | 2 | <0.10–<0.11 | 8 | <0.04–<0.12 |
| p,p′-DDE | 16 | 0.02–0.49 | 10 | 0.35–15.00 | 31 | 0.17–64.00 |
| | 5 | <0.09–<0.33; <4.0 | 0 | | 0 | |
| Dieldrin | 0 | | 0 | | 8 | 0.10–1.20 |
| | 23 | <0.01–<4.00 | 10 | <0.05–<4.20 | 11 | <0.07–<5.30 |
| Heptachlor epoxide | 2 | 0.01 | 0 | | 4 | 0.06–1.10 |
| | 8 | ND[a] | 2 | <0.10–<0.11 | 6 | <0.06–<0.12 |
| Mirex | 1 | 0.01 | 0 | | 3 | 0.08–0.22 |
| | 9 | ND | 2 | <0.10–<0.11 | 7 | <0.04–<0.12 |
| Oxychlordane | 4 | 0.01–0.03 | | | 10 | 0.01–5.21 |
| | 6 | ND | | | 0 | |
| PCB | 0 | | 9 | 0.26–24.00 | 24 | 0.12–52.00 |
| | 21 | <0.31–<1.70; <20 | 1 | <0.56 | 2 | <0.24–<0.62 |
| t-nonachlor | 7 | ND | | | | |
| | 3 | 0.01–0.02 | | | | |
| Aluminum | | | | | 12 | 3–476 |
| | | | | | 7 | <1.58–<8.53 |
| Cadmium | | | | | 0 | |
| | | | | | 12 | <1–<4 |
| Chromium | | | | | 0 | |
| | | | | | 12 | <1–<4 |
| Lead | | | | | 6 | 0.03–0.14 |
| | | | | | 13 | <0.05–<8.0 |
| Mercury | | | | | 18 | 0.06–2.19 |
| | | | | | 1 | <0.10 |
| Selenium | | | | | 12 | 0.53–2.22 |
| | | | | | 0 | |

[a] ND = not detected.

son, Wisconsin. Samples collected in 1991 were tested at Mississippi State University Chemical Laboratory. Quality control/quality assurance procedures at both laboratories are approved by the U.S. Fish and Wildlife Service Patuxent Analytical Control Facility.

For several samples analyzed, particularly blood, contaminant levels were below detection limits (Table 1). When more than half of the samples had levels below the detection limit, we did not statistically analyze the data. When the detection limits were fairly consistent and more than one-half of the samples had detectable values, we assigned a value equal to one-half of the detection limit to samples (S. Wiemeyer, pers. comm.).

Data were transformed to common logarithms ($\log_{10}$) prior to statistical analyses. Transformed data were analyzed using student *t*-tests and analysis of variance (ANOVA) with PC version 6.4 of the Statistical Analysis System (SAS) on a microcomputer. The ANOVA model included year, gender, and an interaction term as independent variables. We used nontransformed data in Pearson product-moment correlations comparing brain and liver levels within the same individual. The significance level for all statistical tests was set at $P < 0.05$. Mean contaminant levels are presented as the arithmetic mean ($\bar{x}$) and the geometric mean (GM).

## Results and Discussion

**p,p′-DDE.** Sixteen of 20 (80%) blood samples analyzed had measurable levels of DDE (Table 1). ANOVA showed no difference in DDE levels by gender ($F = 0.08$, $P = 0.79$), year ($F = 0.13$, $P = 0.73$), or the interaction of gender and year ($P =$

Table 2. Concentrations of p,p'DDE (ppm wet weight) in sharp-shinned hawk tissues from the eastern U.S., 1991–93

| | | BLOOD | | | | BRAIN | | | | LIVER | | | |
|---|---|---|---|---|---|---|---|---|---|---|---|---|---|
| AGE | SEX | *N* | $\bar{x}$ | GM[a] | RANGE | *N* | $\bar{x}$ | GM[a] | RANGE | *N* | $\bar{x}$ | GM[a] | RANGE |
| Adult | | 13 | 0.21 | 0.16 | 0.04–0.49 | 10 | 6.13 | 2.48 | 0.35–15.00 | 19 | 9.98 | 4.56 | 1.04–64.00 |
| | Female | 9 | 0.21 | 0.16 | 0.04–0.49 | 6 | 5.53 | 2.33 | 0.35–14.00 | 10 | 13.90 | 6.37 | 1.50–64.00 |
| | Male | 4 | 0.21 | 0.19 | 0.09–0.35 | 3 | 9.21 | 4.87 | 0.64–15.00 | 7 | 6.77 | 3.95 | 1.04–21.00 |
| | Unknown | | | | | 1 | 0.47 | | | 2 | 1.46 | 1.43 | 1.20–1.72 |
| Juvenile | | 7 | 0.06 | 0.05 | 0.02–0.13 | | | | | 12 | 5.41 | 2.15 | 0.17–23.81 |
| | Female | 5 | 0.05 | 0.05 | 0.02–0.10 | | | | | 5 | 2.79 | 1.19 | 0.17–7.30 |
| | Male | 2 | 0.08 | 0.07 | 0.04–0.13 | | | | | 6 | 8.36 | 4.05 | 1.30–23.81 |
| | Unknown | | | | | | | | | 1 | 0.89 | | |

[a] GM = geometric mean.

0.99) for adult sharp-shins. Therefore, we combined data for adults and compared DDE levels to those found in juveniles (Table 2). Adult levels (GM = 0.16 ppm) were significantly higher than those of juveniles (GM = 0.05 ppm) ($t$ = 3.32, $P$ = 0.004). Similarly, Elliott and Shutt (1993) found that juvenile birds on their first southward migration had significantly lower blood plasma values of DDE than adults. They reported about 0.025 ppm ($N$ = 20) in juvenile samples and about 0.25 ppm ($N$ = 76) in adult samples collected from sharp-shins at Whitefish Point and Hawk Cliff on the eastern Great Lakes. Both sites are farther inland than Hawk Mountain but the breeding and wintering range of sharp-shins trapped at these sites overlaps at least 50% with ranges of birds traveling past Cape May and Hawk Mountain (Duncan 1982, Clark 1985, Struve and Goodrich 1992).

Using the regression equation (DDT in eggs = 6.243 × DDT in blood $^{1.033}$) developed by Henny and Meeker (1981), we determined that the geometric mean level of 0.16 ppm DDE in adult sharpshin blood in our study would result in an estimated 0.94 ppm in eggs (range = 0.40–2.99 ppm). This is lower than the 6.12–9.17 ug/ml reported by Snyder et al. (1973) from four sharp-shinned hawk eggs collected in New York and Pennsylvania and the 5.42–9.12 ppm in three eggs from eastern Canada reported by Meyer (1987). Meyer (1987) also reported that shell thicknesses were 23% below average. Noble and Elliott (1990) reported levels of 3.5–18.6 ppm (GM = 8.3) in 12 sharp-shinned eggs collected in eastern Canada between 1980 and 1988.

In our study, estimated egg values of 0.40–2.99 ppm were much lower than those reported for peregrine falcons (GM = 10.1 ppm) that showed only 13% eggshell thinning and were reproducing well (Ambrose et al. 1988). Fyfe et al. (1988) reported that 1.2–30 ppm DDE in eggs of birds in the genus *Falco* can impair reproduction. For Cooper's hawks (*A. cooperii*), Snyder et al. (1973) suggested that above 3–4 ppm of DDE in eggs was associated with frequent egg breakage. However, Henny (pers. comm.) suggested that accipiters can tolerate higher levels of DDE than those reported by Snyder et al. (1973). Thus, our data suggest that DDE is not impairing sharp-shinned hawk productivity in most of the individuals tested, although we do not know if sharp-shinned hawks respond at the same level of DDE as do these other species of raptors.

We analyzed brain samples from 10 adult sharp-shinned hawk carcasses recovered in the eastern U.S. All had detectable levels of DDE (Table 1) with a GM of 2.48 ppm (Table 2). Sundlof et al. (1986) reported concentrations of 0.25 and 8.50 ppm of DDE in brain tissue of sharp-shins collected in Florida in 1974 and 1977. The highest level detected in our study, 15 ppm, was considerably lower than lethal levels of brain DDE (213–301 ppm) reported in American kestrels (*Falco sparverius*) by Porter and Wiemeyer (1972) and Henny and Meeker (1981).

We analyzed liver samples from 31 sharp-shins for DDE. The highest concentration of DDE detected was 64 ppm in an adult female. DDE levels in adult liver samples were not different by gender (ANOVA: F = 1.66, $P$ = 0.23) or year (ANOVA: F = 0.001, $P$ = 0.96). Similarly, DDE levels in juvenile liver tissues were not different by gender (ANOVA: F = 2.64, $P$ = 0.14) or year (ANOVA: F = 0.54, $P$ = 0.49). The interaction of gender and year was

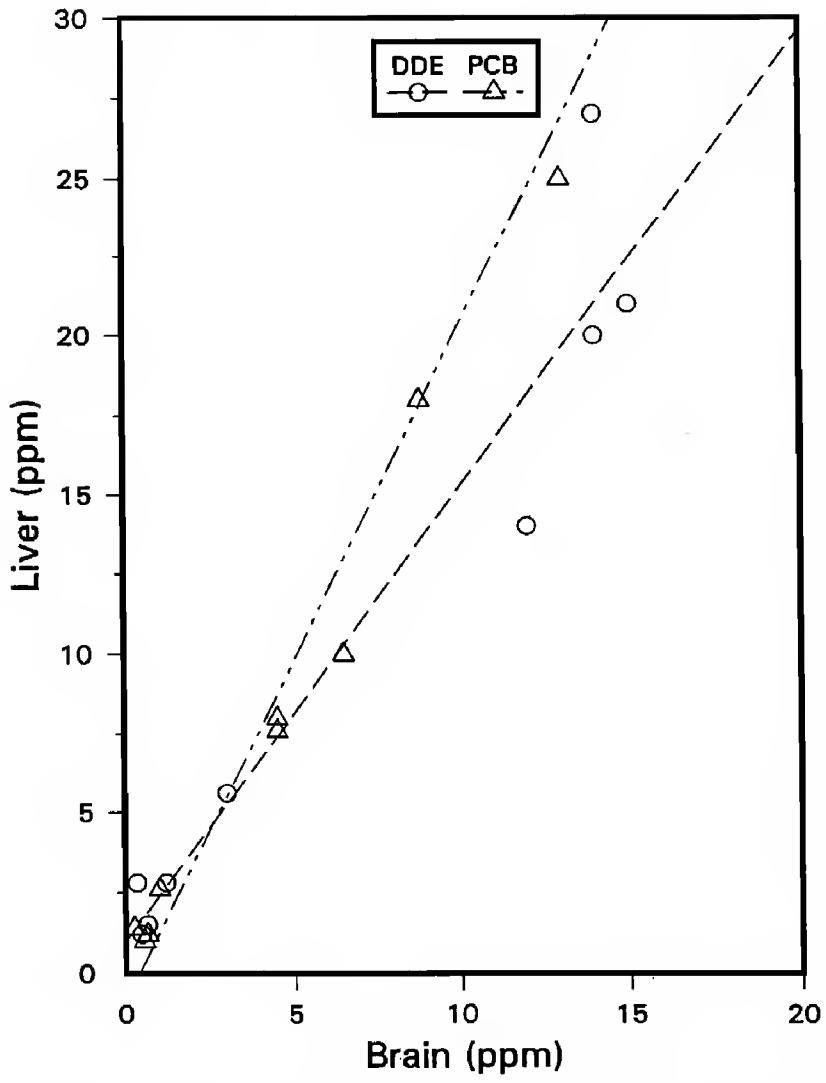


Figure 1. Relationship between eastern sharp-shinned hawk brain and liver tissue for DDE and PCB contaminant levels (ppm).

not significant for either adults ($P = 0.28$) or juveniles ($P = 0.11$). Therefore, we combined data and compared levels in adult and juvenile liver tissue. Although the average DDE level was over twice as high in adult samples (Table 2), the difference was not significant ($t = 1.55$, $P = 0.13$). Small sample size and high variability among samples likely explains this lack of significance. Sundlof et al. (1986) analyzed liver samples from two sharp-shins collected in Florida in 1971 and 1974. DDE concentrations were 0.24 and 2.10 ppm, lower than those detected in our study.

We obtained 10 brain and liver samples from the same carcass. DDE levels in the brain showed a strong and significant correlation to those in the liver ($r = 0.97$, $P = 0.0001$) (Fig. 1).

The minimum level of DDE that affects reproduction in sharp-shins is unknown, thus we do not know if DDE is playing a role in sharp-shinned hawk declines. The estimated egg DDE levels from our study are lower than those reported for earlier years (Snyder et al. 1973, Meyer 1987, Noble and Elliott 1990). In addition, DDE levels detected in birds from the eastern flyway (GM = 0.16 ppm) were higher than those measured in two sharp-shins trapped at Hawk Ridge, Minnesota (GM = 0.035 ppm) (Andersen et al. 1992), a midwestern site where sharp-shinned hawks counted on migration are remaining steady or increasing (Kerlinger 1992). Concurrent studies of contaminant levels and nesting success in eastern sharp-shinned hawks are needed to address this question.

**Polychlorinated Biphenyls (PCBs).** We analyzed 21 blood samples from adult sharp-shinned hawks for PCBs and none contained detectable levels (Table 1). Detection limits for the 11 samples collected in 1993 ranged from 0.31–1.70 ppm with one sample having a detection limit of 20 ppm. Detection limits were high for blood samples due to the small amount of material available for chemical analysis.

Nine of 10 brain samples from adult carcasses collected in the eastern U.S. had detectable levels of PCBs (Table 1). Values ranged from 0.26–24.00 ppm with a GM of 2.65 ppm (Table 3). Sundlof et al. (1986) reported 0.05 ppm PCB in brain tissue from a sharp-shinned hawk collected in Florida. Heinz et al. (1984) reported >300 ppm in brains of birds poisoned by PCBs, levels much higher than those found in our study. Residues in brain appear to be good indicators of PCB stress in birds (Stickel et al. 1984). It is not known, however, what levels of PCBs impair reproduction in sharp-shinned hawks.

PCBs were detected in 24 of 26 liver samples collected in our study (Table 1) with the highest level at 52 ppm (Table 3). There was no significant difference in PCB levels in adults by gender ($F = 0.95$, $P = 0.42$), year ($F = 0.37$, $P = 0.56$), or the interaction of gender and year ($P = 0.66$). Thus, we combined data and compared PCB levels in adults and juveniles. Adult levels (GM = 5.08) were not significantly higher than those in juveniles (GM = 1.6; $t = 1.98$, $P = 0.06$), possibly due to small sample size and high variability among samples.

We analyzed 10 brain and liver samples from the same carcass. PCB levels in brain tissue were highly correlated with those in liver tissue ($r = 0.99$, $P = 0.0001$) (Fig. 1).

PCBs are primarily industrial, not agricultural, pollutants. Thus, PCB residues in birds tend to be high in areas with heavy industrial use or discharge (Fleming et al. 1983, Eisler 1986). Environmental

Table 3. Concentrations of PCBs (ppm wet weight) in sharp-shinned hawk brain and liver tissues from the eastern U.S., 1991–93. Results from blood samples were not included because all values were below detection limits.

| | | Brain | | | | Liver | | | |
|---|---|---|---|---|---|---|---|---|---|
| Age | Sex | *N* | $\bar{x}$ | GM[a] | Range | *N* | $\bar{x}$ | GM[a] | Range |
| Adult | | 10 | 6.35 | 2.65 | 0.26–24.00 | 16 | 10.30 | 5.08 | 0.86–52.00 |
| | Female | 6 | 5.22 | 2.35 | 0.26–13.00 | 9 | 9.35 | 4.66 | 0.86–25.00 |
| | Male | 3 | 10.50 | 5.38 | 1.00–24.00 | 6 | 13.24 | 7.36 | 2.60–52.00 |
| | Unknown | 1 | 0.64 | | | 1 | 1.20 | | |
| Juvenile | | | | | | 10 | 5.79 | 1.61 | 0.12–35.30 |
| | Female | | | | | 4 | 2.90 | 0.95 | 0.12–9.70 |
| | Male | | | | | 6 | 7.72 | 2.27 | 0.31–35.30 |

[a] GM = geometric mean.

contamination resulted from industrial discharges, improper disposal of PCB wastes to municipal sewage treatment plants or landfills and dumps, and especially through atmospheric transport of incompletely incinerated PCBs (Eisler 1986). Freshwater sediment is a major terrestrial reservoir for PCBs (Eisler 1986). Although the levels of PCBs we found in sharp-shinned hawks probably are not affecting the health of these birds, they do seem elevated for terrestrial birds (L. Shutt, S. Wiemeyer, pers. comm.).

**Other Organochlorine Pesticides.** Aldrin was not detected in any of the samples tested from the eastern sharp-shinned hawk population and detection limits were low (Table 1). Cis-nonachlor was detected in two of 10 liver samples tested. Detection limits for the eight remaining liver samples and two brain samples were low. Similarly, heptachlor epoxide was detected at low levels in four of 10 liver samples and two of 10 blood samples with low detection limits for the remaining two blood, six liver, and two brain samples. Mirex was detected in only three of the 10 liver samples and one blood sample; detection limits of the remaining samples were low. Low detection limits of these contaminants indicate that they were present only in low levels, if at all.

All 10 liver samples tested for oxychlordane had detectable levels ranging from 0.01–0.72 ppm (wet wt), with one individual at 5.21 ppm. The lethal hazard zone for brain tissue begins at 5 ppm (Stickel et al. 1979). Acute toxicity has occurred in predatory birds at 3–10 ppm in liver (Cooke et al. 1982). Sublethal effects of this chemical are not known, nor at what levels sublethal effects occur. Although used widely in the past (Eisler 1990), the only current legal use of chlordane in the U.S. is for fire ant control in power plants (Briggs 1992). Thus, sharp-shinned hawks wintering in the southern U.S. could be exposed to this organochlorine compound. With the spread of fire ants across the southern U.S., there is increasing pressure to allow greater use of chlordane. Sharp-shinned hawks are an ideal species for monitoring bioaccumulation of chlordane in terrestrial systems.

Dieldrin was not detected in blood or brain samples (Table 1). Only 8 of 19 liver samples had detectable levels of dieldrin with a mean of 0.35 ppm (GM = 0.25 ppm). Detection limits for dieldrin in brain and liver tissue were sometimes elevated because the PCB signals peaked in the same area of the chromatogram interfering with identification of dieldrin (T. Noltemeyer, Hazleton Environmental Services, pers. comm.).

**Mercury.** Mercury was detected in 18 of 19 liver samples analyzed in this study (Table 1). Mercury levels in adult liver samples (Table 4) collected in 1991 (GM = 0.98 ppm) were significantly higher ($t = 6.20$, $P < 0.0001$) than those collected in 1993 (GM = 0.12 ppm). The samples from each study were analyzed by different laboratories; thus, the difference we saw may be due to differences in methods or equipment sensitivity of the two laboratories. Both levels of mercury, however, were well below levels (>20 ppm) of mercury residues found in tissues of other birds that died of mercury poisoning (Finley et al. 1979). In a review of mercury studies on birds, Ohlendorf (1993) found that <1–10 ppm of mercury in liver was considered a normal background level for birds in general, while >6 ppm was considered toxic.

**Other Metals.** Aluminum was not detected in

any of the 1993 liver samples tested but detection limits were high. Aluminum was found in the 1991 samples ranging from 3–22 ppm with one sample at 476 ppm. Selenium was detected at low levels in 12 liver samples analyzed, while cadmium and chromium were not detected in any of the samples. Low detection limits for these metals indicate that they were present in very low levels, if at all.

Lead was detected in six of seven liver samples collected in 1993 (Table 1) with a mean concentration of 0.07 ppm wet wt (GM = 0.06). The 1991 liver samples had no detectable levels of lead in 12 liver samples; however, detection limits were much higher than those in 1993 ranging from 2.0–8.0 ppm. Lead levels found in our study likely are not detrimental to sharp-shinned hawks. Ohlendorf (1993) reviewed studies that suggested <0.5–5.0 ppm (dry weight) of lead in liver tissues for various bird species can be considered representative of background levels.

## SUMMARY

DDE, PCBs, and mercury were detected most often and in highest concentrations in the sharp-shinned hawks we sampled. No contaminants were present at concentrations that might cause outright mortality with the possible exception of one individual with high oxychlordane residues in the liver. Most samples had low concentrations of contaminants. It is not known, however, at what levels these contaminants might impair reproduction in sharp-shinned hawks. Migration count data (declining sharp-shinned hawk numbers in the East, stable in the Midwest) coupled with contaminant data (higher DDE levels in blood in eastern sharp-shins than in midwestern) do not rule out the possibility that contaminants may be impairing reproduction in some individuals from the eastern population. However, our data are inconclusive regarding effects on the eastern population as a whole because of a lack of concurrent data on reproduction and contaminant levels.

Although our data are inconclusive, they suggest some interesting avenues for further research. Why does the eastern population of sharp-shinned hawks have higher contaminant levels than the midwestern population? How does this terrestrial bird bioaccumulate PCBs, an aquatic contaminant? Further, because of a lack of data for contaminant levels in sharp-shinned hawks, our results provide comparative data for future monitoring of contaminants in these birds.

Elliott and Shutt (1993) postulated that sharp-shins were exposed to pesticide contaminants on southern wintering grounds, both in the southern U.S. and in Central and South America. They found that juvenile birds left the Canadian breeding grounds with low contaminant levels and returned the next spring with the same levels as adults. Similarly, we found lower levels in juveniles than in adults. Banding data from eastern Great Lakes sites indicate that part of their sharp-shin population winters in Latin America, while banding data from the East show that the majority of sharpshins on the eastern flyway winter in southeastern states and rarely leave the continental U.S. (Clark 1985, Struve and Goodrich 1992).

Pesticide use in Latin America continues to have a significant effect on bird populations (Risebrough 1986) as does use of some pesticides (e.g., chlordane) in the U.S. Sharp-shinned hawks feed mainly on small birds, with neotropical migrants comprising about a third of the diet (Storer 1966). Consequently, sharp-shins likely are exposed to contaminants by consuming contaminated prey both on their wintering and breeding grounds. Further study of contaminant levels in sharp-shinned hawks with concurrent research on prey availability and productivity is necessary.

Sharp-shinned hawks may be a good indicator species for monitoring contaminants in terrestrial systems. Recent pressure to increase use of chlordane for fire ant control in the southern U.S. makes it essential to monitor for this contaminant.

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# THE 1990 CANADIAN PEREGRINE FALCON (*FALCO PEREGRINUS*) SURVEY

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ABSTRACT.—In 1990, 24 regions of Canada were searched for nesting peregrine falcons (*Falco peregrinus*). Within the *F.p. anatum* range, 233 sites were occupied: 180 in the Northwest Territories (N.W.T.), Yukon, and Alberta north of 58°N and 32 in southern Canada. The 1990 total of occupied *anatum* nest sites was more than double the number found in 1985–86. *F.p. tundrius* peregrines also increased between 1985–86 and 1990. *F.p. pealei* peregrines on Langara Island were unchanged, but those along the rest of the British Columbia (B.C.) coast increased in numbers. Productivity was high at 1.0–2.9 young per territorial pair, except at the North Slope of the Yukon and Rankin Inlet. With high natural productivity and large releases of captive-raised young in Ontario and Alberta, the peregrine recovery should continue.

KEY WORDS: *peregrine falcon*; Falco peregrinus; *populations*; *survey*; *Canada.*

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Investigación Del Halćon Peregrino Canadiense, Año 1990

RESUMEN.—En 1990, se buscaron los halcones peregrinos (*Falco peregrinus*) en 24 regiones del Canadá. Dentro del terreno *F.p. anatum,* 233 sitios fueron ocupados: 180 en los Territorios del Noroeste, el Yukón, y Alberta más allá de los 58°N y 32 en el sur del Canadá. La suma de los sitios de nidos *anatum* ocupados es más del doble del número encontrado en 1985–86. Los peregrinos *F.p. tundrius* se aumentaron entre 1985–86 y 1990. Los peregrinos *F.p. pealei* mostraron una población estable en lar isla de Langara pero se aumentaron los que están a lo largo del resto de la costa de B.C. (La Colombia Británica). La productividad es fue, de 1.0 a 2.0 crías por pareja territorial, excepto en North Slope del Yukón y Rankin Inlet. Dada la alta productividad natural y el gran número de pájaros criados en cautiverio y puestos en libertad en Ontario y Alberta, la recuperación del halcón peregrino debería continuar.

[Traducción de Peter Imoro y Barbara Krotki]

Falcon enthusiasts and professional biologists conducted peregrine falcon (*Falco peregrinus*) surveys in Canada prior to 1960 and again in 1964–65 (Beebe 1960, Enderson 1965, Fyfe 1969). The 1969 Raptor Research Planning Conference at Cornell University recommended a survey of peregrine falcon nest sites in North America every 5 yr, starting in 1970 (Kiff 1988).

Researchers surveyed peregrines in Canada during 1970 to determine population trends and productivity (Cade and Fyfe 1970) and confirmed the decline that was discussed at the 1965 Madison Peregrine Falcon Conference (Hickey 1969). The 1975 survey further documented declines in the Arctic and boreal forest peregrine populations, both in Canada and the U.S. (Fyfe et al. 1976). During the last coordinated extensive survey of the species' range throughout North America in 1980, peregrine numbers were reduced but stable in northern Quebec, Northwest Territories (N.W.T.), and Yukon. No sites were occupied in the Prairies and only one pair nested in southern Quebec, the boreal forest south of 58°N and east of the Rockies (White et al. 1990). However, nine pairs occupied nest sites in northern Alberta in boreal forest north of 58°N.

While northern and west coast peregrine populations in Canada remained stable or increased in 1985–86, peregrine numbers in the boreal forest south of 58°N and east of the Rockies remained

Table 1. Nest-site occupancy by *anatum* peregrine falcons in Canada during 1990.

| AREA | TOTAL KNOWN SITES[a] | TOTAL KNOWN SITES CHECKED | TOTAL KNOWN SITES OCCUPIED | TOTAL NEW SITES[b] | TOTAL SITES OCCUPIED BY | |
|---|---|---|---|---|---|---|
| | | | | | SINGLE BIRDS | PAIRS |
| Labrador | 25 | 25 | 20 | 1 | 0 | 21 |
| Bay of Fundy | 12 | 11 | 7 | 0 | 2 | 5 |
| Southern Quebec | 10 | 10[c] | 10 | 5 | 3 | 12 |
| Southern Ontario | 35 | 23[d] | 1 | 2 | 1 | 2 |
| Southern Manitoba | 1 | 1 | 1 | 1 | 1 | 1 |
| Southern Saskatchewan | 2 | 2 | 2 | 0 | 1 | 1 |
| Alberta south of 58° | 60 | 60 | 2 | 1 | 0 | 3 |
| Alberta north of 58° | 18 | 18 | 7 | 2 | 0 | 9 |
| Porcupine River, Yukon | 34 | 34 | 30 | 6 | ND[e] | 36 |
| Peel River, Yukon | 31 | 31 | 12 | 2 | ND | 14 |
| Yukon River, Yukon | 36 | 36 | 27 | 6 | ND | 33 |
| Southern Lakes, Yukon | 3 | 3 | 0 | 0 | 0 | 0 |
| Mackenzie Valley, N.W.T. | 107 | 107 | 81 | 7 | 0 | 88 |

[a] Known refers to all sites known before 1990.
[b] New refers to all sites found in 1990.
[c] Plus 135 potential nest sites checked.
[d] Plus 121 potential nest sites checked.
[e] ND = Not determined.

low at eight occupied sites. Six of these eight sites were in cities (Murphy 1990).

Wildlife agencies continued to monitor known nest sites and found new occupied sites since the 1985–86 survey, especially in southern Quebec, Bay of Fundy, northern Alberta, and in cities. More extensive surveys on the Labrador coast identified a significant breeding population (Lemon and Brazil 1990, J. Brazil, pers. comm.). Here, we present the results of the 1990 Canadian peregrine falcon survey and summarize the 5 yr surveys in Canada as well as results from 1965–66 (Hickey 1969).

## METHODS

Remote areas in the Arctic and Labrador were surveyed with helicopters and fixed-wing aircraft. Areas more easily accessible were surveyed by foot and vehicle. Boats and canoes were used to survey some lakes and rivers.

Terminology was a major obstacle to the interpretation of earlier peregrine surveys. We followed Murphy's (1990) definitions of "nest site," the actual site of the nest; "occupied nest site," a nest site where one or two territorial adults were present; "breeding pair," a pair that laid at least one egg; "successful pair," a pair that raised at least one chick to fledging or assumed to have fledged; and "territorial pair," a pair that defended its nesting cliff against other peregrines (Ratcliffe 1980). Most young were counted at 2–4 wk of age and not at fledging. We continued to use the term "historic site" to provide an indication of the intensity of survey effort; however, historic sites are designated as "known" in the tables and represent any sites used prior to 1990.

Single birds are reported separately from pairs since their breeding status was unknown, especially after one survey. Following terminology of Brown (1974) and Postupalsky (1974), we present productivity as the number of young per territorial pair and per successful pair.

The reported number of young or eggs typically reflects the results of a single nest visit regardless of the stage of development of the young. In Labrador and the N.W.T., nest visits coincided with or were near hatching time so only young chicks were encountered and early failed nesters may have been missed. Such single visits, well before fledging, would obviously overestimate productivity (Ellis 1988). In Ungava Bay and Langara Island, nests were visited and chicks banded when they were 20–30 d old. In the Bay of Fundy, southern Quebec, Ontario, Manitoba, Alberta, N.W.T., and parts of the Yukon, nests were visited several times but, often not as late (nestlings 35 d of age) as recommended by Ellis (1988). Therefore, the estimates of young fledged, and hence productivity, are more accurate for these latter sites.

Since a comparable number of sites were surveyed with similar effort as in the previous 5 yr surveys, we compiled population numbers and productivity data for most regions from Enderson (1969), Hickey (1969), Cade and Fyfe (1970), Fyfe et al. (1976), Court et al. (1988), Munro and Van Drimmelen (1988), Murphy (1990), and White et al. (1990).

## RESULTS

The 1990 surveys covered 24 regions of Canada (Tables 1–4), including some regions not surveyed

Table 2. Nest-site occupancy by *tundrius* and *pealei* peregrine falcons in Canada during 1990.

| AREA | TOTAL KNOWN SITES | TOTAL KNOWN SITES CHECKED | TOTAL KNOWN SITES OCCUPIED | TOTAL NEW SITES | TOTAL SITES OCCUPIED BY | |
|---|---|---|---|---|---|---|
| | | | | | SINGLE BIRDS | PAIRS |
| *tundrius* | | | | | | |
| Ungava Bay, Quebec | 58 | 58 | 31 | 3 | 0 | 34 |
| North Slope, Yukon | 17 | 17 | 1 | 0 | 1 | 0 |
| Rankin Inlet, N.W.T. | 26 | 26 | 26 | 0 | 0 | 26 |
| *pealei* | | | | | | |
| Langara Island, B.C. | 7 | 7 | 7 | 0 | 0 | 7 |
| Queen Charlotte Islands, B.C. | 144 | 129 | 71[a] | 0 | 6 | 53 |
| Vancouver & Gulf Islands, B.C.[b] | 39 | 33 | 17 | 1 | 11 | 7 |

[a] Includes 15 sites estimated by extrapolation to unsurveyed areas plus a correction for detection error due to difficulties of gunfire-from-boat survey method on ocean. Not included are 17 singles that showed no defensive behavior or attachment to a site.
[b] Includes 1991 Gulf Islands data because of inadequate survey in 1990.

previously. A total of 631 of 665 (95%) historical peregrine sites were checked for occupancy. A total of 233 sites were occupied within the *anatum* range with the majority (77%) in the N.W.T., the Yukon, and Alberta north of 58°N. Throughout Canada, productivity was 1.0 or more young per territorial pair and as high as 2.9 young, with the exception of Rankin Inlet and the North Slope (Table 6).

**Labrador Coast—J. Brazil, Newfoundland and Labrador Wildlife Division.** From Cape Chidley in the north to Table Bay in the south, 25 known nest sites were checked at least once on either 18–19 June or 7–12 July (Table 1). Some sites were visited incidentally on 28 May, 15 July and 7 August. The Kogaluk River was also surveyed on 8 July, where one new site was located.

Twenty-one sites were occupied, two were unoccupied, and the occupancy at three sites was uncertain. Fifteen of the 21 occupied sites had eggs, young, or both. Nest contents were determined at

Table 3. Productivity of *anatum* peregrine falcons in Canada during 1990.

| AREA | TERRITORIAL PAIRS | SUCCESSFUL PAIRS | TOTAL YOUNG[a] | AVERAGE YOUNG/ TERRITORIAL PAIR | AVERAGE YOUNG/ SUCCESSFUL PAIR |
|---|---|---|---|---|---|
| Labrador | 21 | 13 | 26[b] | 2.6[c] | 3.3[d] |
| Bay of Fundy | 5 | 3 | 6 | 1.2 | 2.0 |
| Southern Quebec | 12 | 9 | 17 | 1.4 | 1.9 |
| Southern Ontario | 2 | 2 | 4 | 2.0 | 2.0 |
| Southern Manitoba | 1 | 1 | 2 | 2.0 | 2.0 |
| Southern Saskatchewan | 1 | 1 | 1 | 1.0 | 1.0 |
| Alberta south of 58° | 3 | 2 | 3 | 1.0 | 1.5 |
| Alberta north of 58° | 9[e] | 5 | 13 | 1.4 | 2.6 |
| Porcupine River, Yukon | 30[f] | 18 | 50 | 1.7 | 2.8 |
| Peel River, Yukon | 12[f] | 9 | 29 | 2.4 | 3.2 |
| Yukon River, Yukon | 27[f] | 19 | 46 | 1.7 | 2.4 |
| Mackenzie Valley, N.W.T. | 88 | 70 | 182 | 2.1 | 2.6 |

[a] Captive-raised young omitted.
[b] Includes only young from pairs where number of young were determined ($N = 8$).
[c] Calculation includes only pairs where number of young and eggs were determined ($N = 10$).
[d] Calculation includes only pairs where number of young were determined ($N = 8$).
[e] Includes one pair in the adjacent N.W.T.
[f] Production of new pairs unavailable.

Table 4. Productivity of *tundrius* and *pealei* peregrine falcons in Canada during 1990.

| AREA | TERRITORIAL PAIRS | SUCCESSFUL PAIRS | TOTAL YOUNG | AVERAGE YOUNG/ TERRITORIAL PAIR | AVERAGE YOUNG/ SUCCESSFUL PAIR |
|---|---|---|---|---|---|
| *tundrius* | | | | | |
| Ungava Bay, Quebec | 34 | 32 | 100 | 2.9 | 3.1 |
| North Slope, Yukon | 0 | 0 | 0 | 0 | 0 |
| Rankin Inlet, N.W.T. | 26 | 8 | 20 | 0.8 | 2.5 |
| *pealei* | | | | | |
| Langara Island, B.C. | 7 | 5 | 14 | 2.0 | 2.8 |

Table 5. Total occupied peregrine falcon nest sites found in selected regions of Canada from 1965–66 to 1990 Number in parentheses indicates pairs present.

| AREA | 1965–66 | 1970 | 1975 | 1980 | 1985–86 | 1990 |
|---|---|---|---|---|---|---|
| *anatum* | | | | | | |
| Labrador | 0 | 2 (2) | 0 | ND | 2 (2) | 21 (21) |
| Bay of Fundy | ND[a] (2) | 0 | 0 | 0 | 1 (1) | 7 (5) |
| Southern Quebec | ND (2) | 0 | ND | 1 (1) | 1 (1) | 15 (12) |
| Ontario | 0 | 0 | 0 | 0 | 1 (0) | 3 (2) |
| Manitoba | ND | ND | ND | 0 | 1 (1) | 2 (1) |
| Saskatchewan | ND | 0 | ND | 0 | 2 (1)[b] | 2 (1) |
| Alberta south of 58° | 8 (6) | 1 (1) | 0 | 0 | 2 (2) | 3 (3) |
| Alberta north of 58° | ND (4) | 2 (1) | 3 (3) | 9 (9) | 6 (5) | 9 (9) |
| Porcupine River, Yukon | ND | ND | 8 (8) | 16 (13) | 14 (11) | 36 (ND) |
| Peel River, Yukon | ND | ND | ND | 18 (12) | 12 (10) | 14 (ND) |
| Yukon River, Yukon | ND | 6 (5) | 6 (5) | 12 (10) | 22 (18) | 33 (ND) |
| Mackenzie Valley, N.W.T. | 14 (ND) | 9 (6) | 24 (21) | 20 (15) | 45 (ND) | 88 (77) |
| *tundrius* | | | | | | |
| Ungava Bay, Quebec | ND | 12 (9) | 11 (9) | 10 (10) | 23 (23) | 34 (34) |
| North Slope, Yukon | ND | ND | 5 (5) | 2 (0) | 0 | 1 (0) |
| Rankin Inlet, N.W.T. | ND | ND | ND | 8 (8) | 26 (ND) | 26 (26) |
| *pealei* | | | | | | |
| Langara Island, B.C. | 9 (6) | 6 (5) | 6 (6) | 6 (6) | 6 (5) | 7 (7) |
| Queen Charlotte Islands, B.C.[c] | 76 (55) | 56 (46) | 60 (51) | 73 (58) | 50 (ND) | 64 (53) |
| Vancouver and Gulf Islands, B.C. | ND | ND | ND | 5 (4)[d] | 13 (10)[e] | 18 (7)[f] |

[a] ND = Not determined.

[b] (1) refers to male peregrine mated with female prairie falcon.

[c] First number is an estimate of occupied sites which includes pairs, singles defending/attached to sites plus, except in 1965–66, an extrapolation to unsurveyed areas based on results of other surveys. A correction for detection error is not included.

[d] Only Gulf Islands data.

[e] Includes one site from Triangle Island (not surveyed).

[f] Excludes Triangle Island.

Table 6. Productivity of peregrine falcons found in selected regions of Canada surveyed every 5 years from 1970–1990. Productivity data indicate average young per successful pair, and in parentheses, average young per territorial pair.

| AREA | 1970 | 1975 | 1980 | 1985–86 | 1990 |
|---|---|---|---|---|---|
| *anatum* | | | | | |
| Labrador | 2.0 (2.0) | 0 | ND | 3.0 (1.5) | 3.3 (2.6) |
| Bay of Fundy | 0 | 0 | 0 | 0 | 2.0 (1.2) |
| Southern Quebec | 0 | ND[a] | 2.0 (2.0) | 0 | 1.9 (1.4) |
| Ontario | 0 | 0 | 0 | 0 | 2.0 (1.3) |
| Manitoba | ND | ND | 0 | 0 | 2.0 (1.0) |
| Saskatchewan | 0 | ND | 0 | 0 | 1.0 (0.5) |
| Alberta south of 58° | 3.0 (1.5) | 0 | 0 | 2.0 (2.0) | 1.5 (1.0) |
| Alberta north of 58° | 0 | 0 | 3.2 (2.1) | 0 | 2.6 (1.4) |
| Porcupine River, Yukon | ND | ND | 1.7 (1.2) | 2.6 (2.0) | 2.8 (1.7) |
| Peel River, Yukon | ND | ND | 0 | 2.3 (1.9) | 3.2 (2.4) |
| Yukon River, Yukon | 2.0 (2.0) | 1.0 (0.4) | 2.2 (1.3) | 2.8 (2.2) | 2.4 (1.7) |
| Mackenzie Valley, N.W.T. | 2.3 (1.4) | 1.3 (0.9) | 2.0 (1.5) | 2.1 (1.7) | 2.6 (2.1) |
| *tundrius* | | | | | |
| Ungava Bay, Quebec | 1.7 (1.3) | 1.8 (1.8) | 2.7 (2.7) | 3.2 (2.7) | 3.1 (2.9) |
| North Slope, Yukon | ND | ND | 0 | 0 | 0 |
| Rankin Inlet, N.W.T. | ND | ND | 3.3 (2.9) | 1.8 (0.6) | 2.5 (0.8) |
| *pealei* | | | | | |
| Langara Island, B.C. | 2.2 (2.2) | 2.4 (2.0) | 2.2 (2.2) | 2.0 (1.6) | 2.8 (2.0) |
| Queen Charlotte Islands, B.C. | 2.5[b] (ND) | 3.2[c] (ND) | 2.5 (2.1) | ND | ND |
| Vancouver and Gulf Islands, B.C. | ND | ND | ND | ND | ND |

[a] ND = Not determined.
[b] Young per 11 successful pairs (Munro and Van Drimmelen 1988).
[c] Young plus 2 pipping eggs per 5 successful pairs (Munro and Van Drimmelen 1988).

10 nests; eight nests contained 26 young and two nests contained four eggs each.

These surveys indicated a reoccupancy rate of 80%, which is likely an underestimate because some females may not have flushed and went undetected. For example, the nest at Cape Kakkiviak appeared unoccupied in mid-June but contained young during the second week of July. Conversely, a nest at Little South Wolf Island, occupied in late May, was abandoned by July.

**Bay of Fundy—B. Johnson, Canadian Wildlife Service.** In New Brunswick, 11 of 12 known nest sites along the Fundy coast, from the lower end of the Bay at Grand Manan to the upper Bay near Sackville, were surveyed on 4 July. In addition, seven cliffs in the lower Saint John River system were also checked because peregrines had been reported nearby. Fundy National Park staff surveyed all of the park's coastline. In Nova Scotia, Five Islands was visited on 2 June but Ile Haute was not checked. The Minas Channel and Basin, from Cape Split near Blomidon, were checked on 5 and 18 July.

Five pairs nested in the Bay of Fundy, New Brunswick (Table 1). Three pairs hatched three young each but only one pair fledged three, another pair fledged two, and the last pair fledged one. Although no pairs were confirmed in Nova Scotia, a single immature bird was seen adjacent to the historic nest site near Cape d'Or. Another single male ("6N2"), a 1983 Cape d'Or release, occupied the Blomidon release site from June through August.

**Ungava Bay, Quebec—D. Bird, Avian Science and Conservation Centre of McGill University.** On 28 and 31 July–2 August, 61 known nest sites were surveyed including three new sites along the Koksoak River, Leaf Bay and the Payne River basins.

Pairs occupied 34 cliffs and 32 sites produced 100 nestlings for an average of 3.1 young per successful pair (Table 4).

These peregrines are subject to human disturbance. One breeding female was shot on Basking Island across from the Payne River settlement. One dead hatchling and an unhatched egg were in the scrape; the male was still present but nondefensive. Occasionally, young were taken as pets by Inuit children along the Koksoak River but the development of "cottages" along this river poses a greater threat to the nesting peregrines.

**Hudson Bay Coast, Quebec—R. Perrault and F. Morneau, Hydro Quebec.** Between 10–20 July, no occupied peregrine nest sites were found during surveys of the Hudson Bay coast west of the Coats River, inland from the mouth of the Grande Riviere de la Baleine and the Petite Riviere de la Baleine, the islands in the Strait of Manitounuk, part of the Islands of Nastapoka, west of Lake Guillaume Delisle, and the mouth of the Nastapoka River. However, later that summer, three birds were seen in the Strait of Manitounuk as well as one adult and two juveniles at Lake Guillaume Delisle indicating that at least one unknown pair successfully raised young in the area.

**Southern Quebec—M. Lepage, Quebec Ministere du Loisir, de la Chasse et Pêche.** All 10 known and 135 potential peregrine nest sites were visited repeatedly by B. Blais and Y. Pinsonneault from 16 May–11 July. Of 12 territorial pairs, nine fledged 17 wild young and five captive-raised young. Three pairs nested in cities (Table 3).

**Southern Ontario—I. Bowman, Ontario Wildlife Branch.** Between 24 April–13 July, 23 of 35 known sites and 121 potential nest sites were visited. Observations occurred at each site for about 1 hr. Pairs occupied two sites; one site had been occupied annually since 1986 and the second site was only 10 km away. A single adult occupied the third site. All adults were unbanded and limited reproductive data were collected indicating 2.0 young per territorial pair (Table 3). An unconfirmed report exists of nesting peregrines in another area.

**Manitoba—R. Nero, Manitoba Wildlife Branch.** J. and P. Duncan conducted an aerial survey of 3438 km of the Hayes, God's, Nelson, Seal, and Churchill rivers, the Hudson Bay coast between the Churchill and Seal Rivers, and the coast by York Factory, from 12–14, 17, and 22 July and 2 August. This survey coincided with the late nestling period and any early failed nesting attempts were missed. No peregrines were sighted but 33 suitable cliff sites were documented.

The only pair of peregrines in southern Manitoba fledged two young in Winnipeg. In addition, a male ("1X"), from the 1989 Winnipeg nesting, occupied a territory at the University of Manitoba in Winnipeg after mid-May.

**Southern Saskatchewan—L. Oliphant, Saskatchewan Cooperative Falcon Project.** Between 7 May–15 June, the South Saskatchewan River from the Alberta border to the Qu'Appelle Dam was surveyed but no peregrines were found. An adult male peregrine occupied a territory at Snake Bite Coulee and courted a female prairie falcon (*Falco mexicanus*) in April. This male also courted a female peregrine flown by a falconer.

One pair of peregrines nested in Saskatoon and fledged three young, two of which were captive-raised (Table 3). Single peregrines were observed in Regina but were presumed to be on passage because none stayed past the end of May.

**Alberta South of 58°N—S. Brechtel, Alberta Fish and Wildlife Services.** All known nest sites in southern Alberta (Court 1993) along the Red Deer, Oldman, Bow, and Milk rivers and some cliffs in the foothills were surveyed. No occupied nor new peregrine nest sites were located. Insufficient resources precluded complete coverage of the foothills region.

Two pairs nested in Edmonton and Calgary. The Edmonton pair laid five eggs, two of which were crushed in the nest. The remaining three eggs were artificially incubated and replaced with two captive-raised young, one of which was injured at fledging. An additional nonbreeding pair spent much of the summer in southern Edmonton. The Calgary pair laid four eggs, hatched three young, and fledged two.

Another pair was reported west of Calgary, but efforts to confirm its presence were unsuccessful. Therefore, this report was treated as an unconfirmed territorial pair pending further surveys because of its proximity to a confirmed 1989 nest site that was unoccupied in 1990.

**Alberta North of 58°N—D. Moore, Alberta Fish and Wildlife Services and J. Dixon, Canadian Parks Service.** In and near Wood Buffalo National Park during May–June, nine territorial pairs were located. Seven pairs were at known sites and two at new sites. Four nests were outside the park: one in the N.W.T. and three on Lake Athabasca. Each pair laid four eggs and 25 young hatched (69%). Ten

captive-raised young from Canadian Wildlife Service, Wainwright, were used to increase brood sizes to four young. Twenty young fledged for an average of 1.4 young per territorial and 2.6 young per successful pair. Four nests failed to fledge young. Three were depredated and one cavity collapsed killing three young.

**Yukon Territory—D. Mossop and G. Mowat, Yukon Department of Natural Resources.** Five subpopulations of peregrine falcons are recognized in the territory. Although the fidelity of peregrines to former nest sites is recognized, all habitat between established pairs was surveyed. Most sites were visited only once during the brood rearing period; however, the Yukon River group was visited repeatedly from incubation through late brood-rearing to maximize the likelihood of determining the presence of newly established and unsuccessful pairs.

Peregrines occupying the Yukon North Slope are considered *F.p. tundrius* (Mossop 1988). In the late 1970s, this population experienced a steep decline and the last productive pair was observed in 1979 (Mossop and Ryder 1980). Annual visits continued since 1979 and the first breeding pair with three eggs was located in 1989 at a site occupied in 1979. A single adult occupied this site in 1990.

In the far northern edge of the boreal forest in the Porcupine River drainage, birds identified as belonging to the interior race or *anatum* (Mossop 1988) declined in the late 1960s and early 1970s. The remnant population showed the first documented recovery in the northwest (Hayes and Mossop 1982). In 1990, we surveyed 34 known sites and located six new sites (Table 1). Reoccupancy was estimated to be 88% and 60% of all known sites produced young. All six new pairs were also productive (Tables 1 and 3). The population is now at its known pre-decline density and apparently still expanding.

In the northeast portion of the boreal forest region in the Peel River drainage and probably communicating directly with the Mackenzie Valley population, there is a subpopulation that declined in the 1960s. Its recovery has been relatively slow and its productivity has been low compared to the other *anatum* subpopulations in the central and north-central portion of the territory (Mossop and Baird 1985).

Only one small segment of this drainage, near the Dempster highway, has been monitored annually. In 1989, we conducted a major survey which gave the data on the current status when combined with the 1990 survey. Adult peregrines occupied 39% of known sites; 75% of pairs at known sites produced young (Tables 1 and 3). Two new pairs were productive. This population is likely at its pre-decline density and is still expanding.

The longest-known subpopulation of peregrines in the Yukon River drainage, that communicates directly with the Alaska Yukon River basin subpopulation, declined through the early 1970s, and by 1978 contained only one occupied nest site. It has exhibited a strong and sustained recovery since that time (Mossop and Baird 1985). In 1990, we resurveyed 36 known sites and found six new sites (Table 1). Peregrines occupied 27 (75%) known sites, with 19 (70%) pairs producing young (Tables 1 and 3). Five new sites were also productive. The population is at least as large as it was originally and continues to expand.

The Southern Lakes area, which includes the large lakes of the southern Yukon, is assumed to be a continuation of the interior habitats of B.C. It was not monitored regularly and the few known breeders disappeared in the 1970s. The three known historic sites remained unoccupied in 1990. Until a more extensive survey is conducted, this group must be considered locally extirpated.

**Mackenzie Valley, N.W.T.—C. Shank, N.W.T. Wildlife Management Division.** The 1990 survey covered the Mackenzie Valley from 80 km upstream of Fort Norman to Inuvik. K. Hodson surveyed the Mackenzie River from 15–26 July. L. Wakelyn and S. Matthews of the Department of Renewable Resources and R. Owens of Foothill Pipe Lines Ltd. surveyed peregrine sites in habitat adjacent to the Mackenzie River from 8–12 June and flew a productivity survey from 14–18 July. A total of 107 known nest sites were surveyed and seven new sites were found (Table 1). Of the known sites, 81 (76%) were reoccupied. Productivity (2.1 young per territorial pair and 2.6 young per successful pair) was the highest recorded for the Mackenzie Valley to date (Tables 3 and 6). Occupancy and productivity of Mackenzie Valley peregrines improved significantly during the last decade.

**Central Arctic Coast, N.W.T.—C. Shank, N.W.T. Wildlife Management Division.** During 1–3 July, approximately 4000 km$^2$ near Coppermine were surveyed by L. Wakelyn, A. Gunn and C. Shank. Occupancy by at least one adult was noted at 61 nest sites. Mean clutch size was 3.4 ($N = 23$) and mean brood size was 2.4 ($N = 5$).

C. Shank surveyed approximately 2000 km$^2$ east

of Bathurst Inlet between 11–17 July. Single adults were seen at 34 nest sites but eggs and young were counted in only few nests. These populations increased dramatically in the past 7–8 yr (Shank et al. 1993).

**Belcher Islands, N.W.T.—J. Nishi, University of Alberta.** Cade and Fyfe (1970) reported nesting peregrines on the Belcher Islands during the 1940s and 1950s, but researchers failed to locate peregrines on the adjacent mainland during the 1975 survey (Fyfe et al. 1976). During 1990, while conducting surveys of plant communities, approximately 20% of the land area of the Belcher Islands was searched from 15 June–10 August. The survey was not systematic nor was it structured to search for peregrine falcons. Nevertheless, three occupied peregrine nest sites were found. Two nests found on 18 July were on 30 m cliffs on isolated rock islands and the third nest containing three young was found on 1 August on an accessible rock ledge on a large rocky bluff. Local Inuit hunters knew of the first two nests and said they saw falcons occasionally during summer.

**Rankin Inlet, N.W.T.—T. Duncan and M. Bradley, University of Saskatchewan.** Nest sites were surveyed from 15 May–15 August within 20 km of Rankin Inlet as part of an ongoing, intensive population study initiated in 1981. Pairs occupied 26 nest sites and only 19 pairs laid 64 eggs. Thirty-nine young hatched and at least 20 fledged from eight nests.

Laying dates and weather conditions indicated that weather did not delay the 1990 breeding season. However, a three-day rain during late July strongly affected the 1990 production. Only 20 of the approximately 39 young hatched survived this storm. Weather appears to be a major factor limiting productivity at Rankin Inlet during at least three of nine years of the study, two of which were successive national peregrine surveys (Bradley 1989, Court et al. 1988).

**Frobisher Bay, N.W.T.—C. Shank, N.W.T. Wildlife Management Division.** L. Wakelyn and P. Kilabuk surveyed Frobisher Bay as far south as Wiswell Inlet and Newell Sound on 5–7 July. Thirty-two peregrine nest sites were checked, some of which may be alternate nest sites. Single birds and pairs occupied 11 nests. Since females were still incubating, no attempt was made to count eggs.

**Thelon Wildlife Sanctuary, N.W.T.—C. Shank, N.W.T. Wildlife Management Division.** During two surveys of known nest sites, 25 sites were found to be occupied. Mean clutch size was 3.0 ($N = 8$) and the mean brood size was 3.3 ($N = 9$). Each nest was visited only once during July or August.

**Nahanni National Park Reserve, N.W.T.—S. Meggs, Canadian Parks Service.** The First Canyon area along the South Nahanni River was surveyed on 29 May. The cliffs from Dry Canyon to Yohin Lake were surveyed twice, once along the rim and then at a lower elevation approximately halfway down the canyon walls. We saw no peregrines nor did any park staff or visitors. In 1985, two pairs of peregrines were observed in the park but were not reported in Murphy (1990). In July 1985, a pair with one young was seen at a cliff nest in First Canyon and in August 1985 another pair with three young was seen approximately 8 km southeast of the first pair.

**Langara Island, B.C.—R.W. Nelson, Camrose, Alberta.** All seven known peregrine nest sites on Langara Island, the northwestern island of the Queen Charlotte Islands, were surveyed from 6–15 June. Seven territorial pairs were found but two had no eggs or young. The five successful pairs produced 14 nestlings, slightly above average for the island (Nelson 1990).

**Queen Charlotte, Vancouver and Gulf Islands, B.C.—W.T. Munro, B.C. Wildlife Branch.** Coastal populations of *F.p. pealei* in B.C. were surveyed on the Queen Charlotte Islands from 21 May–1 June. A total of 59 occupied territories were found. By adding the unsurveyed portion, the number of occupied territories was estimated to be 71.

There were an estimated 10 occupied territories on northern Vancouver Island from a 17 June survey; a single bird also flew from a possible new site. Triangle Island was not checked in 1990 but in 1989 it had seven occupied territories. On the Gulf Islands, three known nest sites were found occupied during a 15 June survey, but a minimum of four sites were assumed occupied because another site appeared to be occupied. This survey was considered incomplete and therefore was repeated in 1991. In 1991 the Gulf Islands shorelines were surveyed on 27–28 May and more inland sites were surveyed on 4 June. The combined 1991 surveys resulted in the location of seven occupied nests.

## DISCUSSION

The *Anatum* Peregrine Falcon Team coordinated the 1990 survey and selected target areas to search for nesting peregrines. The team's objective was to resurvey known areas with an effort similar to that

used in previous 5 yr surveys. The approach taken was to survey historic sites within the *anatum* peregrine range every 5 yr and to encourage annual surveys and more detailed monitoring of selected populations (Erickson et al. 1988). In the Arctic, inadequate resources precluded the resurveying of all areas surveyed between 1970–80. Also, the *tundrius* subspecies had increased in most areas that were being monitored annually, so the need for extensive surveys was not as great (Bromley 1991). Other major gaps in the 1990 survey were in central Quebec, northern Ontario and Saskatchewan, and interior B.C.

The number of occupied *anatum* nest sites increased in most regions of Canada from 1985–86 to 1990 and was higher than at any other time. However, peregrine numbers increased substantially along the Porcupine and Yukon rivers and increased slightly along the Peel River. These increases in the north reflected the improved reproduction and survival of peregrines after the ban of DDT in North America and a decline of DDE in the prey of peregrines (Peakall et al. 1990). Counts in the Mackenzie Valley, Rankin Inlet, and central Yukon were also higher than in previous counts but these increases may be a product of increased survey effort. Likewise, on the Labrador coast and Ungava Bay, population increases since 1985 were most likely due to more intensive survey work (Lemon and Brazil 1990). The northern Alberta and Wood Buffalo National Park population has shown a gradual increase, which is expected because of some human assistance through the provision of additional young.

Peregrines have failed to reoccupy the southern Yukon. Given this increase in peregrine numbers in the Yukon, adjacent Alaska (Ambrose et al. 1988), and N.W.T., repopulation is expected to occur naturally.

Peregrine falcon populations in southern Canada appeared to be recovering. In 1990, 17 pairs nested in southern Quebec and New Brunswick, up from two pairs in 1985 (Murphy 1990). Two pairs nested in southern Ontario and five pairs nested on the Prairies, all in cities. This recovery was due to the release of large numbers of captive-raised young (Holroyd and Banasch 1990). All peregrine pairs that nested on the Prairies in 1990 were in cities and the number of pairs increased from four to five since 1985 (Murphy 1990). In Quebec and Bay of Fundy, a dramatic population increase occurred because of large releases of captive-raised young, but in Ontario the recovery was slower. The scarcity of occupied peregrine nest sites in Ontario fails to reflect the large number of released captive-raised young. Releases occurred at widely scattered locations from Ottawa to Thunder Bay from 1977–90 (Holroyd and Banasch 1990). Apparently, many single falcons returned but never mated and others returned, but to Quebec City, Boston, Toledo, and Winnipeg.

Population trends of *tundrius* were not as clear because of the lack of detailed population data prior to 1980. The increases noted in the number of territorial pairs in the Rankin Inlet population from 1980–85 were primarily the result of more intensive surveys.

Intensive annual monitoring of Langara Island's *pealei* peregrines has shown a stable population with healthy reproduction since 1968 (Nelson 1990) and the population appears to be at capacity in view of the once plentiful but now diminished food supply (Nelson and Myres 1975). Extensive 5 yr surveys of the Queen Charlotte, Vancouver, and Gulf Islands indicate a stable population on the B.C. coast.

The objective of the *Anatum* Peregrine Falcon Recovery Plan is 10 breeding pairs in six of nine management zones by 1992. This goal was reached in three zones: southern Quebec-New Brunswick-Nova Scotia-Prince Edward Island, the Yukon and the Mackenzie Valley. In southern Ontario and the Prairies, the goal of 10 pairs was partly achieved. The number of pairs is unknown in central Quebec, northern Ontario and interior B.C.

Although peregrines showed stable population numbers prior to the DDT era (Hickey 1969), post-DDT populations increased along with the release of captive-raised young and declining DDT levels. In Canada, southern populations reestablished and increased due to the introduction of captive-raised young while northern populations increased with little human intervention and because of lower DDT levels. When annual population growth rates were calculated between 1980, 1985, and 1990, a general pattern emerged. In the north, populations increased annually by 13% in Ungava, 16% in Mackenzie Valley, and 7% in the Yukon. In the south, the populations increased by 72% in southern Quebec and 50% in New Brunswick and Nova Scotia from 1985–90. Thus, populations in certain areas of large releases of captive-raised young increased by 50% or more per year, while populations where few or no releases occurred grew at

16% or less per year. These figures paralleled the number of peregrines in the eastern U.S. which increased at 26% per year with releases and were projected to increase at up to 5% per year without (Grier and Barclay 1988).

The observed productivity of the nesting pairs in most areas was adequate to support continued increases across the range of the peregrine. Newton (1979) stated that stable peregrine populations produced 1.0–1.5 young per nesting pair per year on average. Grier and Barclay (1988) projected that with an average production of 2.5 young per year per successful pair and 66% of nesting pairs being successful (1.68 young per territorial pair per year and no supplemental young), that the eastern U.S. peregrine population would increase at the rate of 0.3–4.5% per year. In 1990, the peregrines surveyed in Canada produced greater than 1.5 wild young per territorial pair except in the Bay of Fundy, southern Quebec, southern Saskatchewan, Alberta south of 58°N, the North Slope and Rankin Inlet.

Although productivity is only one part of any population model and does not itself indicate a population increase (Grier 1979), the consistency of productivity over 1.5 young per territorial pair per year indicates that peregrines throughout much of Canada produced at a rate that should sustain or increase their numbers where they occur in sufficient density. In Ontario and on the Prairies, where the number of pairs is low or the recovery slow, the risk of extirpation from stochastic events indicate a need for the continued release of captive-raised young. In addition, five nesting pairs and six of seven occupied nest sites on the Prairies in 1990 were in cities, not on prairie cliffs.

Nisbet (1988) and Peakall (1990) commented on the relative lack of success of the release program in Canada compared to the eastern U.S. However, they based their comparisons on the number of pairs in 1985, not single birds, which were more numerous than paired birds in the areas of release in southern Canada (Holroyd and Banasch 1990). Also, the number of pairs and singles more than doubled through 1988 (Holroyd and Banasch 1990) and doubled again by 1990. The return rates for captive-raised released young falcons are lower in Canada than in the U.S. (Holroyd and Banasch 1990). The establishment by 1990 of 19 pairs and six single falcons in southeastern Canada and five pairs and two singles in the Prairies indicates that breeding pairs were successfully established in most areas of releases.

Throughout Canada, productivity was 1.0 or more young per territorial pair and as high as 2.9 young, with the exception of Rankin Inlet and the North Slope. Low productivity at Rankin Inlet was due to a single rainstorm and did not reflect production in recent years (Court et al. 1988). A single adult occupied a nest site in the North Slope.

Court et al. (1988) criticized 5 yr surveys as too infrequent to accurately monitor population changes in most populations north of 60°N because low production and a decrease in the number of successful pairs may simply represent normal fluctuation rather than a major population decline. At temperate latitudes, however, other authors have noted the relative stability of peregrine populations, especially when unaffected by pesticides or recovering from them (Ratcliffe 1980, Newton and Mearns 1988, Nelson 1990). Since the peregrine is not an ephemeral species (Galushin 1974, Hunt 1988), annual surveys of its entire range are unnecessary to monitor changes in its population size.

A related issue is the effectiveness of single visits to monitor breeding peregrine populations. The single surveys in the N.W.T., Yukon, Ungava Bay, and Labrador occurred at hatching or during the nestling stage. Adult peregrines are more aggressive at this stage than during incubation and thus more likely to be detected. However, any failed nesters are less likely to be detected by later surveys. Mark Bradley (pers. comm.), who monitored the Rankin Inlet population throughout the entire 1986 breeding season, determined that less than 10 of the 26 original pairs remained territorial after nest failure in early incubation caused by the spring snowstorm. Thus, a single survey after such a nest failure would have underestimated the population by 60%. The effect of catastrophic weather was noted by Court et al. (1988) and in Labrador during 1989 and 1991 by J. Brazil (pers. comm.). Thus, the results from single surveys should be used with caution since they could underestimate population size and productivity.

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# SHORT COMMUNICATIONS

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## GOLDEN EAGLE PREDATION ON PRONGHORNS IN WYOMING'S GREAT DIVIDE BASIN

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KEY WORDS: *pronghorn; golden eagle;* Antilocapra americana; Aquila chrysaetos; *predation; behavior; Wyoming; Great Divide Basin*

From 1980–85, while studying pronghorn antelope in Wyoming's Great Divide Basin, we recorded seven incidents of golden eagles (*Aquila chrysaetos*) attacking pronghorns (*Antilocapra americana*). Most reported observations of golden eagle predation on pronghorns involve newborn fawns during spring and summer (Beale and Smith 1973, Barrett 1978, Beale 1978, Bodie 1978, Von Gunten 1978, Autenrieth 1980), but a few attacks have been recorded in winter (Lehti 1947, Thompson 1949, Bruns 1970, Goodwin 1978) when two adult males, one adult female and four fawns were killed. Herein, we provide further evidence for golden eagle predation on pronghorns, particularly during the winter.

The study was conducted in the Great Divide Basin, an area of shrubsteppe habitat located northwest of Rawlins, Wyoming. Golden eagles utilize the area preying on pronghorns, white-tailed jackrabbits (*Lepus townsendii*), desert cottontails (*Sylvilagus audubonii*) and sage grouse (*Centrocercus urophasianus*) (U.S. Dept. of Interior 1978).

We observed eagles in the study area throughout the year. Rabbits and sage grouse were observed almost daily. In late summer 1982, we conducted two strip-transect surveys to estimate lagomorph densities. These surveys were conducted by driving a vehicle at night and counting all lagomorphs in a transect defined by the width of the headlight beam. In March 1983, we attempted to count all pronghorns and eagles in the 182 $km^2$ area where we observed eagle attacks. Counts covered the entire area and were made using a Piper Supercub aircraft flown at an elevation of 60 m along 400-m-wide north-south transects. Surveys began 30 min after sunrise on days when visibility was excellent and used one observer who always looked away from the sun.

During our 1983 survey, we counted 3230 pronghorn and 23 golden eagles for a density of 17.7 pronghorn and 0.13 golden eagles per $km^2$. Data from our two strip-transects provided an estimate of 0.94 lagomorphs per ha. We considered these estimates to be accurate indicators of the relative abundance of predators and prey in our study area.

All but one of our observations of golden eagles attacking pronghorns were made from November through February 1981–84. Six observations were made from a vehicle and one from the air. We do not know if our aircraft influenced pronghorn or eagle behavior during this attack, but during all five winter observations made from the vehicle, we were parked, watching groups of pronghorns and they began to run from eagles, not from us. We were traveling along a two-track road when our summer observation was made and, because pronghorns were running when we first observed them, we do not know if our vehicle or the eagle first caused pronghorns to flee. Eagles were not marked, thus we do not know how many individual eagles were involved in attacks or if the same eagle may have been observed in more than one attack.

Each winter attack involved a single golden eagle and groups of 120 to 350 pronghorns. The pronghorns became alarmed when the eagle vocalized while circling a herd at an estimated elevation of 60 m. Pronghorns fled only when the eagle flew low (10 m or less above the ground) and directly toward them. In one instance, pronghorns fled when the eagle was seen flying at them from 300 m away. When running from an eagle, pronghorns appeared to group more closely than when we observed them escaping from coyotes (*Canis latrans*) or humans.

When circling pronghorns, eagles did not always attack. When they did attack, there was a consistent pattern of circling, vocalizing, and flying away from the group of pronghorns just prior to the initiation of a chase. During all five winter observations made from the ground, eagles vocalized. Pronghorns watched eagles fly away and continued to watch the area where they had disappeared. When an eagle reappeared flying low to the ground and directly toward them, the pronghorns bunched more closely together and then ran. When the pronghorns

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fled, the eagle targeted the last animal in the group, and all subsequent attacks were directed toward that isolated individual.

All attacks were similar and, after the eagle caught up to the pronghorn, it landed on its back at points varying from just posterior to the withers to slightly anterior to the white rump patch. Once an eagle landed on a fleeing pronghorn, the pronghorn fell or continued to run apparently trying to dislodge the bird. The longest time an eagle rode a pronghorn was approximately 20 sec for an estimated distance of 200 m. Eagles did not balance well on fleeing pronghorns and all attacks ended when the eagle and pronghorn fell to the ground. Prior to hitting the ground, the eagle folded its wings and after the fall, it remained standing on the ground for approximately 30 sec before attacking again. During each encounter we observed, the eagle attacked the same pronghorn from one to four times and, of the three pronghorns killed, one, two, and three attacks were involved before the pronghorn was killed. Kills we witnessed involved one female and one male fawn (estimated age 8–9 mo) and one male (estimated age 20–21 mo). Pronghorns that escaped did so by running back to the fleeing herd. We do not know the fate of these animals, but we did observe otherwise healthy looking pronghorns with dried blood on their backs.

Eagles began feeding on pronghorns after they fell. Two pronghorns appeared paralyzed, possibly from spinal injuries, but remained alive for at least 10 min after eagles began feeding. Eagles always began feeding dorsally on the carcass at the point where their talons had punctured the skin. Kills were quickly detected by other golden eagles and coyotes. On one occasion, a second eagle arrived within 8 min after a kill was made and, after 27 min, five golden eagles were feeding on the kill. We observed coyotes approaching golden eagles feeding on dead pronghorns, but eagles did not attack coyotes, nor did coyotes displace eagles from the carcasses.

We observed a fawn attacked by an eagle in July 1985. In this instance, approximately 10 pronghorns were running in front of our vehicle when a fawn (<1 mo. old, estimated weight 4 to 5 kg) in the rear of the group was attacked by a pursuing golden eagle. The eagle grasped the fawn in the back, lifted it approximately 10 m vertically then released its grip dropping the fawn. It rose, began to run with an awkward gait and the eagle initiated a second stoop, but appeared to shy from our vehicle. The fawn ran out of sight, and the eagle flew in the opposite direction.

During winter, pronghorns became concentrated in our study area which may, in part, explain why we observed more eagle predation in winter than in summer. Based on our field observations of jackrabbit, cottontail rabbits and sage grouse, these prey appeared to be available. Thus, we doubt that eagles were attacking pronghorns because alternate prey was unavailable. More likely, they were related to the winter conditions which made pronghorns vulnerable to eagle predation. Snow depth ranged from 0–10 cm and did not impede pronghorn movement nor did it appear to influence eagles when they attacked pronghorns.

It was interesting that eagles prey on adult pronghorns nearly as frequently as on immatures. Nearly 50% of our observations involved attacks by single eagles on adults and fawns from the previous spring when they approach adult body size. Bruns (1970) observed a similar trend toward winter predation on adults by single eagles hunting herds of pronghorns. Tandem hunting has been reported in breeding areas (Collopy 1983), but this behavior is apparently rare in winter (Tjernberg 1986). Collopy (1983) reported two golden eagles preying on black-tailed jackrabbits (*Lepus californicus*) and smaller mammals. Hatch (1968) observed a pair of golden eagles successfully killing a red fox (*Vulpes fulva*). In both reports, one eagle either flushed the prey or diverted its attention while the second eagle attacked. Thompson (1949) observed two golden eagles simultaneously chasing two separate pronghorn herds but the size and relative conspicuousness of pronghorns may eliminate the need for tandem hunting of this species.

Bruns (1970) reported an eagle attacking two different pronghorns before successfully killing a female fawn. Eagles we observed attacking pronghorns directed their attention to a single animal and actually rode on their victims for as long as 20 sec. Bruns (1970) observed an eagle riding a pronghorn for nearly 5 min.

We observed a 50% success rate for golden eagles preying on pronghorns in winter. This estimate is higher than the 23–30.5% success rate reported by Collopy (1983) for eagles hunting small mammals in Idaho, and the 21% success rate for golden eagles hunting small animals in Sweden (Tjernberg 1986). Small mammals may be better able to find cover for their escape. Pronghorns are more conspicuous than small mammals and they live in open habitats where opportunity to use cover for escape is limited.

Pronghorns seem to recognize eagle hunting behavior and bunch tightly together when they flee from attacking eagles (Bruns 1970). Eagles we observed elicited this response by either vocalizing while circling immediately above pronghorns, or by flying close to the ground directly at herds. This behavior isolates an animal from the group that then becomes the focus of ensuing attack

RESUMEN.—Estudiamos a *Antilocapra americana* en la gran cuenca divisora de Wyoming, en un período de 57 meses, entre 1980 y 1985. Durante este tiempo, registramos siete ataques de *Aquila chrysaetos* sobre *A. americana*. Seis ataques en invierno involucraron un macho adulto, uno del año, una hembra adulta y tres cervatillos. En verano sólo se observó un ataque sobre un cervatillo. En invierno, las águilas mataron el 50% de las veces que atacaron, matando un macho y una hembra (ocho a nueve meses de edad) y un macho de 20 a 21 meses de edad. El

ataque realizado en verano sobre el cervatillo, no fue exitoso, posiblemente debido a nuestra presencia. En invierno, las águilas atacaron a *A. americana* desde la parte posterior de grupos que huían, aterrizaban sobre sus espaldas, derrivaban el grupo o liberaban sus garras antes que *A. americana* cayera. El mayor tiempo de persecusión fue de 20 segundos. Cuando *A. americana,* se sentía amenazada por *A. chrysaetos,* corrían asociados en rebaños, una conducta que difiere de respuestas de escape suscitadas por otro tipo de peligros. Sugerimos que esta respuesta conductual única de *A. americana* puede indicar una interacción histórica depredador-presa.

[Traducción de Ivan Lazo]

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# SEX AND AGE CLASSES OF MIGRATING RAPTORS DURING THE SPRING OF 1994 AT EILAT, ISRAEL

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KEY WORDS: *raptors; 1994 spring survey; sex; age; Eilat.*

Raptor migration counts have been conducted in the spring and autumn from 1977–87 near the northern tip of the eastern arm of the Red Sea at Eilat (Christensen et al. 1981, Shirihai 1987, 1988, Shirihai and Christie 1992, Shirihai and Yekutiel 1991). The most recent survey was conducted in the spring of 1994 (Yosef 1995). Because early reports did not present results pertaining to the sex and/or age classes of the raptors observed on migration, I am presenting the sexes and ages of raptors that passed this observation point during the daily count.

Observations were made at three points for approximately 12 hr/d from 15 February–19 May 1994. Except for two d when observations were terminated due to sandstorms, 92 d of observations were carried out (Yosef 1995). At the observation point, soaring birds frequently flew within 50 m of the observers in mornings and late evenings. Species not aged or sexed were not included in the analyses (e.g., short-toed eagle [*Circaetus gallicus*], booted eagle [*Hieraaetus pennatus*], osprey [*Pandion haliaetus*]), as were individuals that were not identified to the species level.

A total of 1 022 098 raptors of 29 species were counted (Yosef 1995). Of these, only 3993 birds were sexed and/or aged (0.4% of total). Most were steppe eagles (3048,

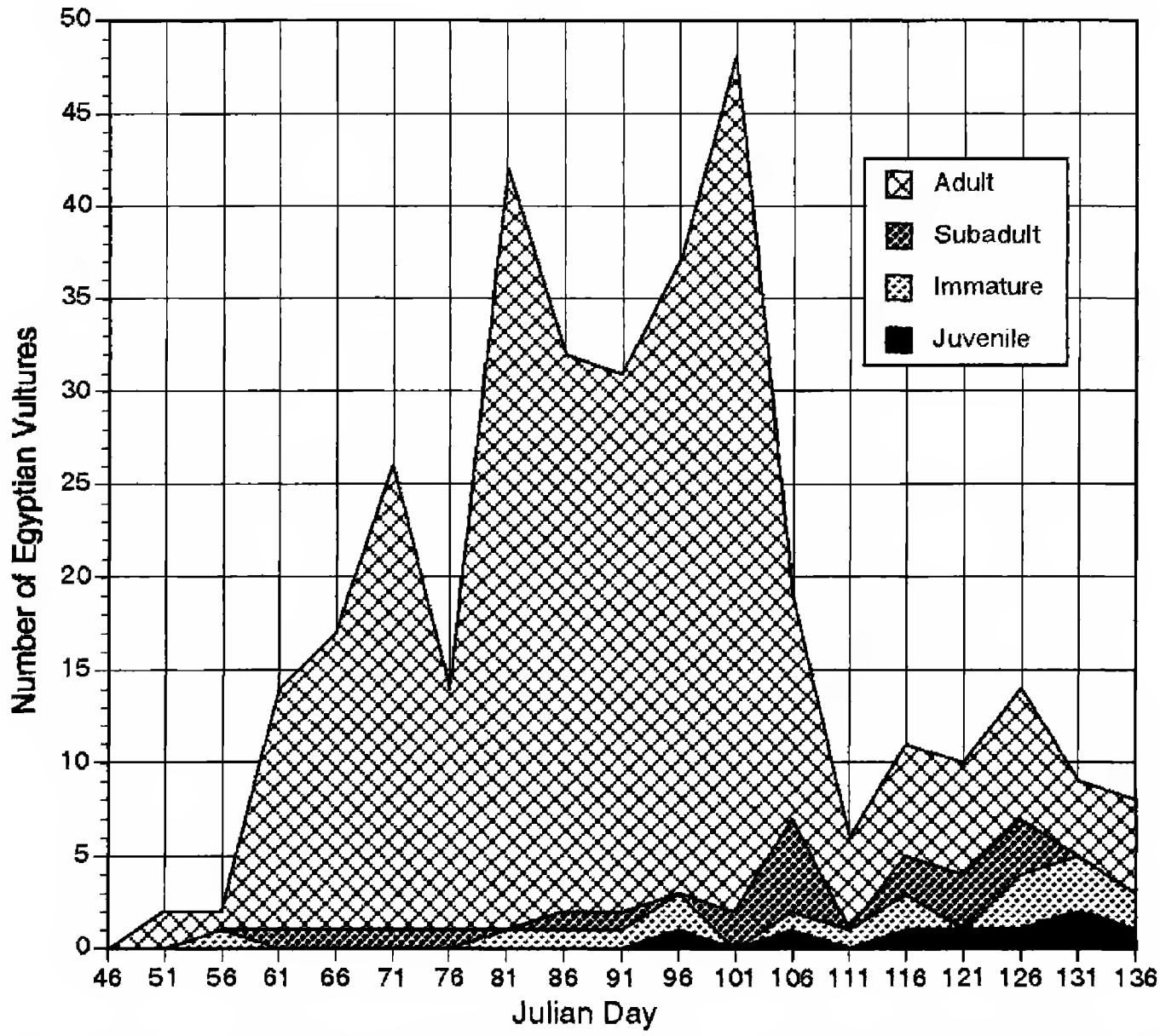


Figure 1. Age classes of migrant Egyptian vultures (*Neophron percnopterus*, $N = 342$) at Eilat in the spring 1994. Data are presented in 5-d periods.

Table 1. Ages and sexes of raptors observed in the spring 1994 at Eilat, Israel. The total observed for each species is given for comparison of percent identified (Yosef 1995).

| | AGE | | | | SEX | |
|---|---|---|---|---|---|---|
| SPECIES | JUVENILE | IMMATURE | SUBADULT | ADULT | FEMALE | MALE |
| Black kite | 1 | | | 4 | | |
| Egyptian vulture | 8 | 18 | 21 | 295 | | |
| Griffon vulture | 1 | | | 13 | 14 | 14 |
| Marsh harrier | | | | | 68 | 19 |
| | | | | | (Juv- 12 | 3) |
| Pallid harrier | | | | | 12 | 40 |
| Montagu's harrier | 1 | | | | 3 | 2 |
| Sparrowhawk | | | | | 12 | 5 |
| Levant sparrowhawk | | | | | 15 | 32 |
| Steppe buzzard | | 8 | | 188 | | |
| Long-legged buzzard | | 6 | | 15 | | |
| Lesser spotted eagle | | 2 | | 2 | | |
| Spotted Eagle | | | | 2 | | |
| Steppe eagle | 243 | 110 | 357 | 2338 | | |
| Imperial eagle | 28 | | 6 | 7 | | |
| Golden eagle | 3 | | 2 | 2 | | |
| Bonelli's eagle | 2 | | | 8 | | |
| Lesser kestrel | | | | | 5 | 9 |
| Eurasian kestrel | | | | | 11 | 22 |
| Red-footed falcon | 1 | | | | 2 | |
| Hobby | | | | 22 | | |
| Eleonora's falcon | | | | 7 | | |
| Sooty falcon | | | | 5 | | |
| Peregrine falcon | | | | 4 | | |
| Barbary falcon | | | | | 1 | 2 |

*Aquila nipalensis*), Egyptian vultures (342, *Neophron percnopterus*), and steppe buzzards (196, *Buteo buteo vulpinus*). Days on which individual raptors or small flocks were observed gave the greatest accuracy in determining sexes and/or ages of migrants. On days when flocks of thousands were counted, only individual raptors that were conspicuous were either aged or sexed. Because migration is species-specific at Eilat with a few species comprising between 65–98% of the birds seen on any given day (Yosef 1995), exceptional species are more conspicuous to the observer (Yosef 1995). In the Eilat region, the two dominant soaring species are the honey buzzard (53.5%, *Pernis apivorus*) and steppe buzzard (37.4%). Because they appear in large flocks, the number of individuals successfully aged or sexed was low (0.05%).

Egyptian vultures (342; 82.0%) were identified as either juveniles, immatures, subadults, or adults (Mundy et al. 1992, Table 1). Adults (295, 86.3%) were observed throughout the survey, but the largest numbers were observed between 4 March–17 April. The bulk of the subadults migrated from 6 April–11 May (Fig. 1). Twenty-one (6.1%) were subadults, 18 (5.3%) immatures, and eight (2.3%) juveniles. Shirihai and Christie (1992) also found that during spring 1985 almost 95% of Egyptian vultures were adults and that nonadults passed mainly in early May. By coalescing data from five separate surveys, Mundy et al. (1992) found that more juvenile/immature (brown) birds fly south to Africa in autumn in comparison to those returning to Europe and Asia in spring. In the 1994 survey, pied birds (adults/subadults) also comprised 95% of the migrating Egyptian vultures indicating that juvenile and immature mortality may be as high as 80% of that age group, or that only a few birds fly back to Europe and Asia while others remain in Africa (Mundy et al. 1992).

Marsh harriers (*Circus aeruginosus*) were observed in small numbers throughout the survey but there were two peaks in the number of harriers passing the observation point between 22 March–10 April and 21–27 April when as many as five harriers/d were observed. Females dominated the migration (68 vs 19) and males were concentrated between 22 March–15 April. During the first peak nine males vs 25 females were observed, but the second peak comprised mostly females (one vs 19). Pallid harriers (*C. macrourus*) were only seen for three wk. Sexes differed in time of migration with males migrating (19

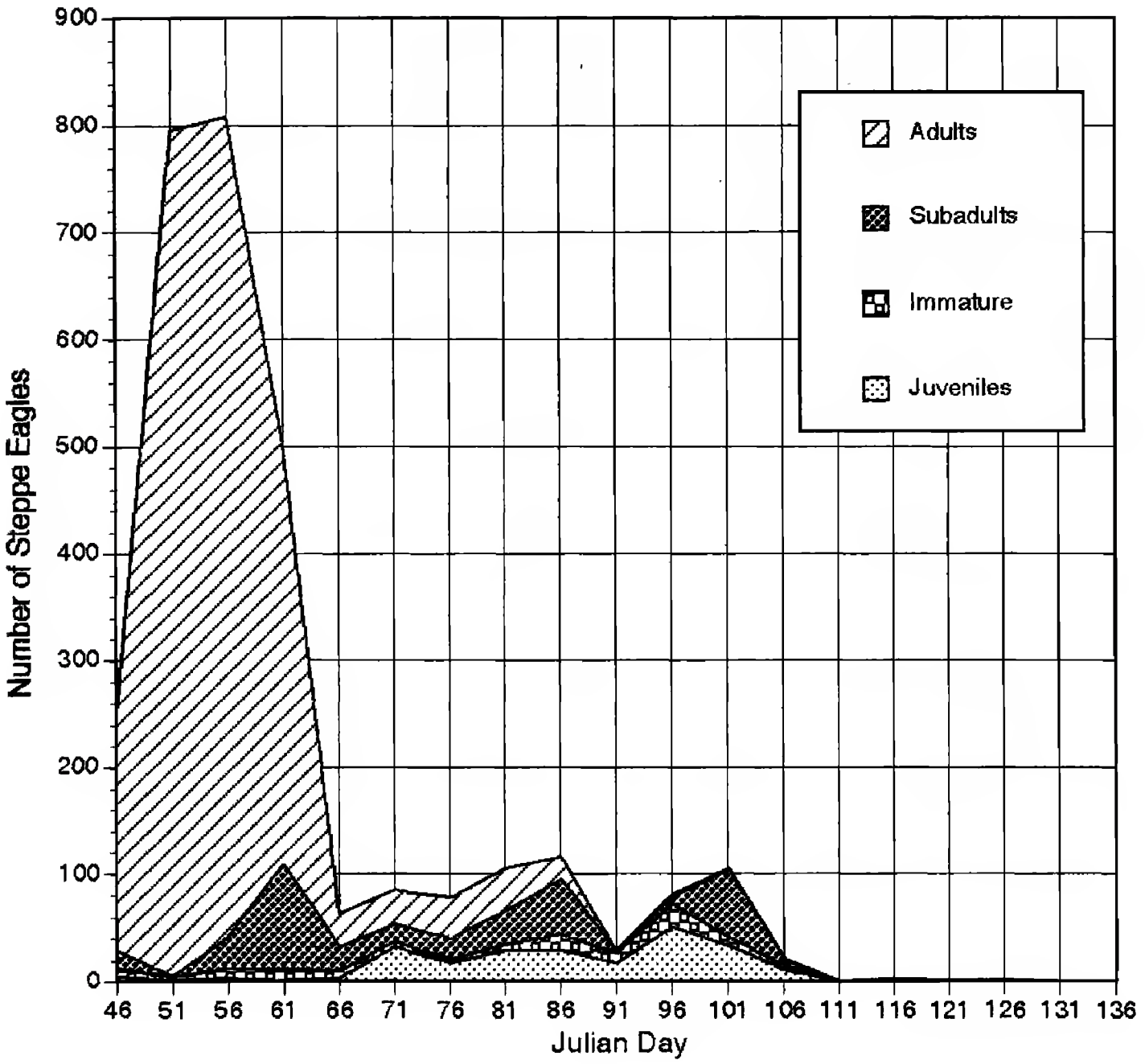


Figure 2. Age classes of migrant steppe eagles (*Aquila nipalensis*, $N = 3048$) at Eilat in the spring 1994. Data are presented in 5-d periods.

March–15 April, median = 26 March) earlier ($t = 4.69$, df = 11, $P < 0.0007$) than females (26 March–10 April, median = 31 March). Montagu's harrier (*C. pygargus*) migrated one wk later than pallid harriers from 8–23 April.

Sparrowhawks (*Accipiter nisus*) were solitary migrants and their size and flight at low altitudes made them difficult to identify. Thus, only 17 (13.9%) of 122 seen were sexed. Nevertheless, females dominated the passage (12 vs five).

Levant sparrowhawks (*Accipiter brevipes*) usually passed through the Eilat mountains in large flocks during the hottest hours of the day and were usually very high. If they roosted, they left before first light so they were difficult to sex and age. This problem was further confounded by the fact that recent radar studies indicate that it is possible that levant sparrowhawks are also nocturnal migrants (Stark and Liechti 1993). A total of 32 males and 15 females were identified mostly when they were in flocks of 5–10 birds.

The steppe buzzard was the second-most-numerous species but only 196 (0.05%) were aged (Table 1). All adults on the 1994 survey were identified between 15 February–26 March. This finding concurred with Shirihai and Christie (1992) who contend that adults predominate in the migration up to mid-April, and then the flight is comprised mainly of juveniles.

Long-legged buzzards (*B. rufinus*) migrated in small numbers from late-February to late-April. Numbers, and consequently age ratios, may have been underestimated because of its similarity to the more numerous steppe buzzard (Shirihai and Forsman 1992). The majority identified were adults.

Although the lesser spotted eagle (*A. pomarina*) was seen in small numbers from mid-March to mid-April, of the 65 observed in 1994, only four were aged (two subadults, two adults).

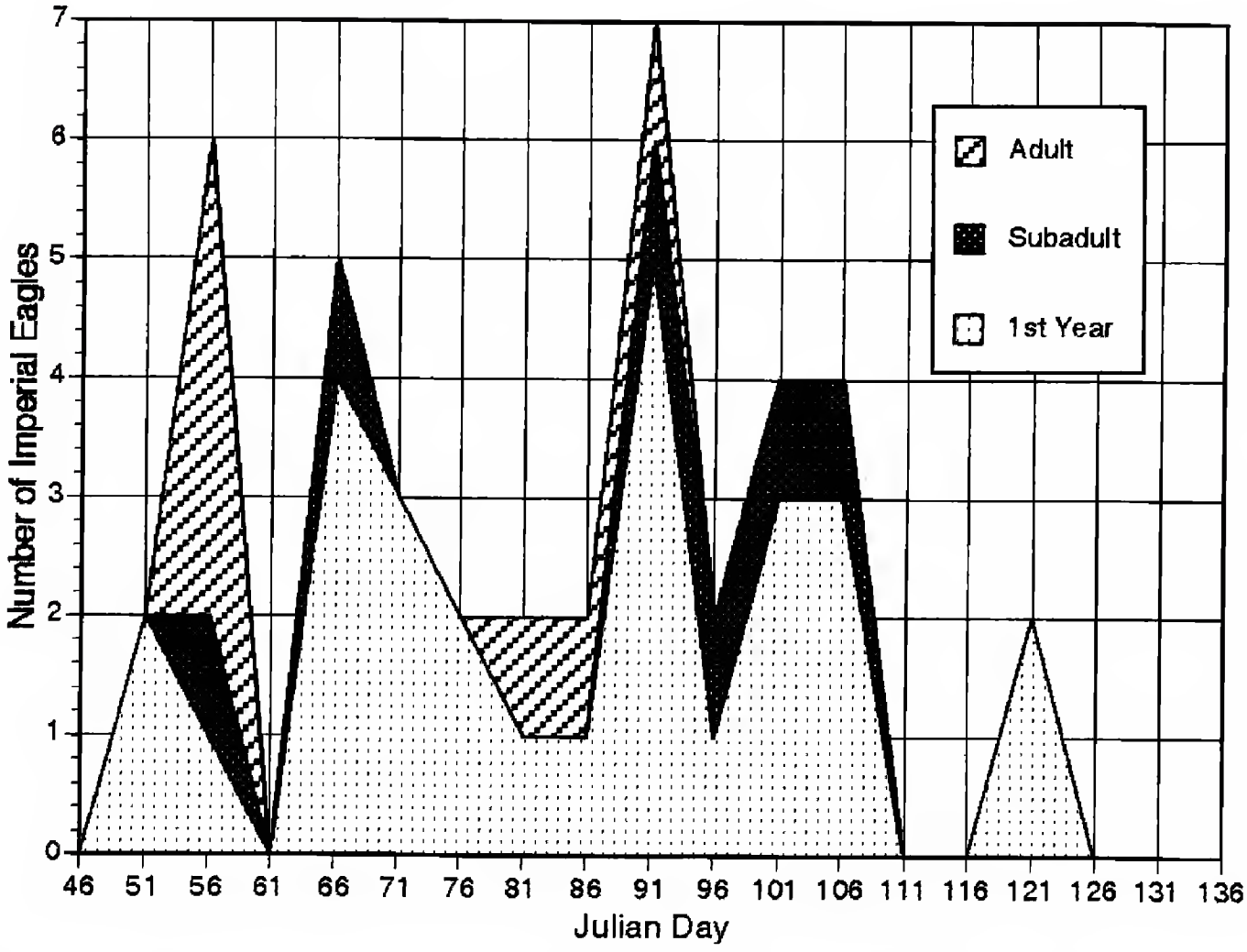


Figure 3. Age classes of migrant imperial eagles (*Aquila heliaca*, $N = 41$) at Eilat in the spring 1994. Data are presented in 5-d periods.

Most adult (2338, 76.7%) steppe eagles migrated by late-March. This concurred with previous surveys which indicated that 60–75% of the steppe eagles migrating through Eilat in spring are of breeding age. Juveniles (1st and 2nd calendar yr) and immature (3rd–5th calendar yr) birds were observed evenly distributed throughout February, March and April (Fig. 2) which differed from Shirihai and Christie (1992) who found that from the end of March and throughout April many immatures pass, while by mid-April the majority are juveniles.

Imperial eagles (*A. heliaca*) migrated throughout the survey. Two peaks previously described (first wave in late-February to mid-March, second wave in early-April, Christensen et al. 1981, Shirihai and Christie 1992) were not evident in 1994 and adults were not predominant. In fact, first-yr birds dominated the migration in general (68%, Fig. 3). Although Shirihai and Christie (1992) implied a decline in numbers between 1977–88 ($r^2 = 0.665$, $N = 6$, $P < 0.048$), this was not substantiated during the 1994 survey ($r^2 = 0.27$, $N = 7$, $P < 0.23$).

Falcons (*Falco* spp.) were spread out over the migration in small numbers. Only 14 (16.9%) lesser kestrel (*Falco naumanni*) were identified and many may have been missed because they migrate mostly along the coast and in open areas. Male kestrels (*F. tinnunculus*) outnumbered females two-to-one. All hobbys (*F. subbuteo*) seen were adults. Many were probably missed owing to their low, dodging flight in the canyons below the observation posts. All seven Eleonora's falcons (*F. eleonorae*) and five sooty falcons (*F. concolor*) seen were also adults. Of the four peregrine falcons (*F. peregrinus*) seen, one was a juvenile and the other three adults. Of the four Barbary falcons (*F. pelegrinoides*) observed, one was a female and the other two males.

Differences between this study and prior surveys in the ages and sexes of migrants observed is indicative of the need to be cautious of using the results obtained from raptor migration surveys. A good example of the potential for error is the very low number of adult imperial eagles observed during this survey compared to 1992 results. It is possible that the majority of the adult population followed routes further north of Eilat, or might even have wintered north of Eilat (western Negev desert and Hula Valley). This study stresses the importance of surveys that identify age and sex of migrant raptors.

RESUMEN.—Varios conteos de rapaces en migración han sido realizados en primavera y verano en Eilat, extremo norte del brazo este del Mar Rojo. Los estudios iniciales no indicaban sexo y/o clases de edad de las rapaces observadas. Durante 92 días de observación, registramos un total de 1022098 individuos de 29 especies de rapaces. De ellos, 3993 individuos fueron sexados y/o clasificados por edad (0.4% del total). La mayoría (3049) correspon-

dió a *Aquila nipalensis,* 342 a *Neophron percnopterus* y 196 a *Buteo buteo vulpinus.* Aquellos días en los que se registraron aves individuales o pequeñas bandadas, fueron los mejores para determinar sexo y/o edad. Este trabajo enfatiza la importancia de los estudios migracionales que identifican sexo y determinan edad de las especies cuando es posible. Información de este tipo es necesaria para obtener estimaciones gruesas de poblaciones continentales de rapaces.

[Traducción de Ivan Lazo]

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# NUMBERS AND DISTRIBUTION OF RED-FOOTED FALCON (*FALCO VESPERTINUS*) NESTS IN VOIVODINA (NORTHERN SERBIA)

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KEY WORDS: *red-footed falcon*; Falco vespertinus; *Voivodina.*

The breeding range of red-footed falcon (*Falco vespertinus*) in the Palearctic extends across the broad band of steppe, foreststeppe and cultivated lands in the north temperate zone (Cade 1982). The western limit of this species extends to the western border of Hungary in Central Europe (Keve and Szijj 1957, Lohmann 1962, Glutz et al. 1971, Cramp and Simmons 1980). Although a relatively large red-footed falcon population occurs in the Carpathian Basin (Keve and Szijj 1957, Glutz et al. 1971, Cramp and Simmons 1980), very little is known about their number and breeding distribution in the northern province of Serbia, Voivodina (Antal et al. 1971, Pelle et al 1977).

Based on limited information, Vasić et al. (1985) made a preliminary estimate of 80 pairs of red-footed falcons in Voivodina. However, this preliminary work did not provide a clear picture of the distribution of nesting areas. Herein, I provide the results of a recent survey of the numbers and distribution of red-footed falcon nests in the area.

Voivodina is a mostly flat region in northern Serbia that lies in the southeastern part of the Carpathian Basin. It is divided by the Danube, Tisa, and Sava Rivers into three areas: Bachka, Banat, and Srem (Fig. 1). The Bachka region (8956 km$^2$) lies between the Tisa and Danube Rivers and borders with Hungary in the north. The Banat region (8886 km$^2$) is north of the Danube River and east of the Tisa River and borders with Hungary and Romania. The Srem region (3838 km$^2$) lies between the Danube and Sava Rivers and borders with Croatia. The area ranges from 70–200 m above sea level and two mountains, the Fruška Gora (539 m) in Srem and the Vršac Mountains (641 m) in southeastern Banat rise above the large plain. They support deciduous forests dominated by oak (*Quercus* spp.), linden (*Tilia* spp.), and hornbeam (*Carpinus* spp.). The lower slopes have mostly been cleared and are used for pastures, vineyards and orchards. Voivodina is largely agricultural and only 5.4% is forested. The most common soil type is chernozem, a black soil, covering 60% of the arable land. It is extremely fertile and mainly used for cultivation of crops such as wheat, maize, sugar beets, sunflowers, and soya. Industrial crops, fodder crops and vegetables are also cultivated on the black marsh soil. Alluvial soils occur in river valleys and support willows (*Salix* spp.) and poplar (*Populus* spp.) forests and meadows. Approximately 10% of Voivodina has saline soils which are used for pastures and, in some places, fishponds.

The red-footed falcon census was performed from 28 April–14 July in 1990, and from 27 April–22 July in 1991. In 1990, surveys were conducted for 28 d, while in the second year the survey took 33 d. Srem, Bachka, and Banat were surveyed on 7, 18 and 36 d, respectively. In both years, an observer and I drove all the main roads in Voivodina, usually 200–400 km per day, for a total of >20 000 km. Over a 3-d period at the end of April, and over 16 d in May in both survey years, I also searched for rook colonies and recorded any red-footed falcons observed.

Horváth (1955) observed red-footed falcons laying eggs in the second half of May and concluded that nests became occupied 2–3 weeks before egg laying. To determine clutch and brood sizes of red-footed falcons, nest trees were climbed twice in June and July. Only those nests containing at least one egg during the first nest visit were included in nest size estimates. When nests were impossible to climb, they were checked twice from the ground to make certain that the female was in the same nest and to count nestlings standing on the edge of the nest.

To make the census more complete, information on rook colonies and locations of red-footed falcon nests was obtained from the Association for Protection and Study of Birds of Voivodina. This was originally reported by A. Zsulyevits, M. Dević and J. Rašajski (Table 1).

I found 308 pairs and 124 pairs of red-footed falcons in Voivodina in 1990 and 1991, respectively (Table 1). Breeding pairs were found only in Bachka and in Banat (Fig. 1). In 1990 four, while in 1991, five pairs nested in northwestern and southeastern Bachka between the Danube and Tisa Rivers. In all cases, red-footed falcons nested in isolated magpie (*Pica pica*) or hooded crow (*Corvus corone cornix*) nests.

In Banat, 304 pairs of red-footed falcons were found in 1990, while only 119 pairs were recorded in 1991. Most nests were either along the Tisa River or in the foothills of the Vršac Mountains (Fig. 1). In 1990, red-footed falcons nested in 15 separate locations. With the exception of three isolated pairs found in old magpie nests (in the vicinity of the villages of Bočar, Dobrica and Konak), red-footed falcons nested in rook (*Corvus frugilegus*) colonies.

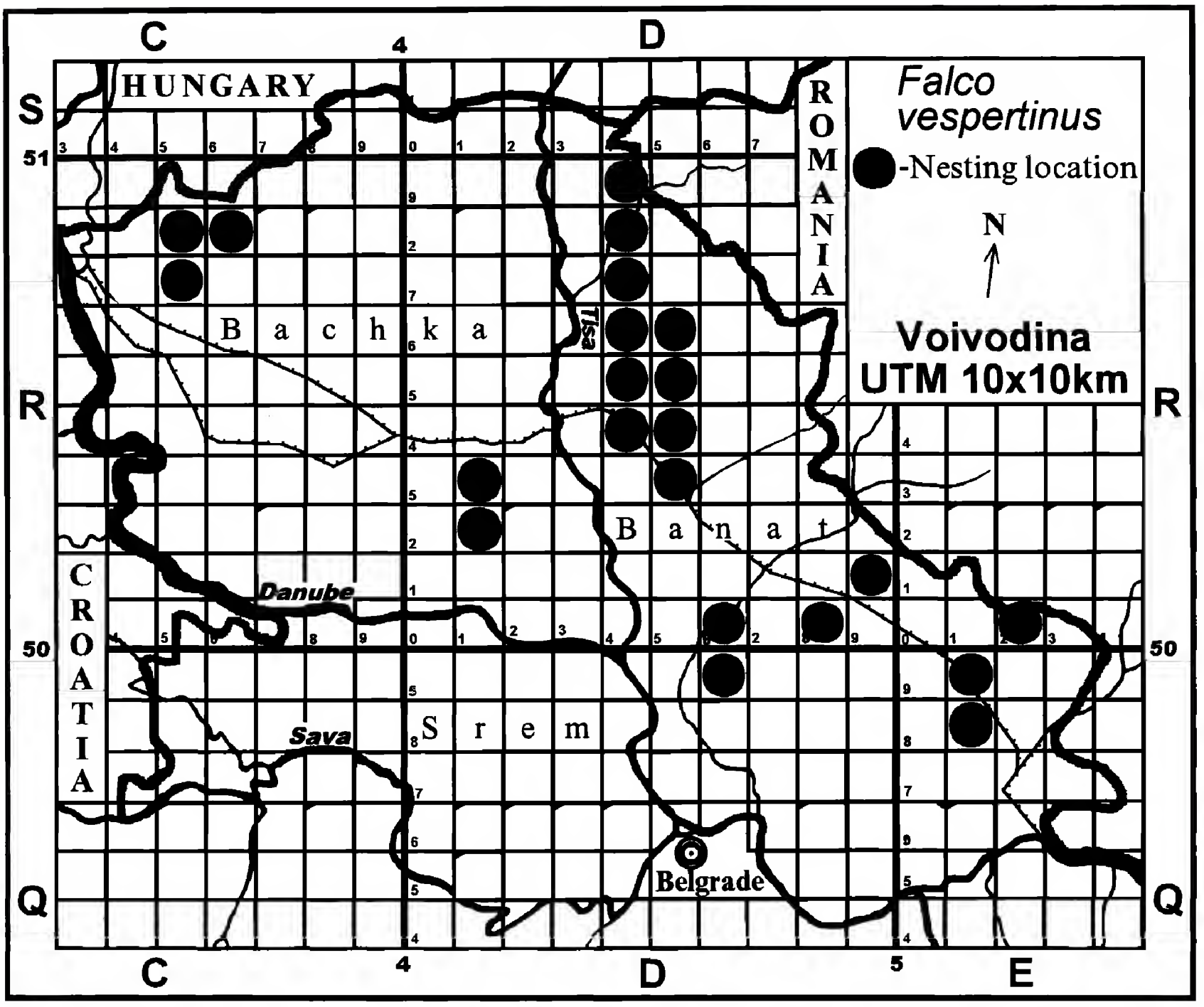


Figure 1. Breeding distribution of red-footed falcon in Voivodina during 1990 and 1991 based on the bird census data.

Five large colonies were used including one near the village of Jazovo where 147 pairs of red-footed falcons nested. In 1991, 19 sites were used in Banat. At 9 sites, 11 pairs nested in abandoned magpie or hooded crow nests. At the other 10 sites, 108 pairs nested in rook colonies. The four largest colonies were near Jazovo (29 pairs), Melenci (27 pairs), Vlajkovac (16 pairs), and Torda (14 pairs) (Table 1).

Birds of prey that breed in colonies tend to nest in the same location year after year (Newton 1979). In this study, red-footed falcons also reused colonies but the numbers of breeding pairs decreased from 308 pairs in 1990 to only 124 pairs in 1991. There are several possible reasons for the difference between the results of the two census years. The census in Voivodina was conducted near the periphery of the species distribution. Peripheral populations tend to react more sensitively to environmental changes than do more central populations (Udvardy 1969). The spring of 1991 was cooler and wetter than 1990, which delayed nesting of rooks. Thus, when red-footed falcons arrived at the end of April to breed, rooks were still occupying nests and they were forced to find alternative nest sites. It is possible that some of the falcons may have gone further north, but the rest may have used abandoned magpie and hooded crow nests or tried to occupy empty nests inside rook colonies that were still active. Usually, red-footed falcons that nest in colonies are more reproductively successful than solitary nesters (Haraszthy and Bagyura 1993). Unfavorable weather conditions may, therefore, lower reproductive success of red-footed falcons by forcing them to nest in less suitable, isolated magpie and hooded crow nests.

Raptors may also become concentrated in the breeding season in areas with abundant food (Newton 1979). Voivodina is predominantly agricultural and grasslands

Table 1. Number of breeding pairs of red-footed falcon found in Voivodina during 1990 and 1991.

| UTM | LOCALITY | 1990 | 1991 |
|---|---|---|---|
| | Bachka | | |
| CR57 | Lenija, 6 km northeast of Sombor (A. Zsulyevits) | 1 | 2 |
| | Milčić, 8 km northeast of Sombor (A. Zsulyevits) | 1 | — |
| CR58 | Rančevo, 14 km north of Sombor (A. Zsulyevits) | — | 2 |
| CR68 | 5 km southwest of Aleksa Šantić | — | 1 |
| DR12 | 2 km west of Gospodjinci | 1 | — |
| DR13 | 8 km northeast of Temerin | 1 | — |
| | Banat | | |
| DR44 | 7 km north of Melenci | 3 | — |
| | 5 km northeast of Melenci | 46 | 27 |
| DR45 | Slano kopovo, 9 km east of Novi Bečej | — | 1 |
| | 6 km southwest of Bašaid | — | 2 |
| DR46 | 5 km west of Bočar | 1 | — |
| | 6 km southwest of Novo Miloševo | 6 | — |
| DR47 | 1 km east of Idjoš | — | 2 |
| DR48 | 2 km southeast of Jazovo | 147 | 29 |
| DR49 | 2 km southwest of Vrbica | — | 1 |
| | 2 km west of Banatski Monoštor | — | 1 |
| | 1 km west of Banatski Monoštor | — | 1 |
| DR53 | 3 km northwest of Jankov Most | 1 | 2 |
| DR54 | 2 km south of Torda | 20 | 14 |
| | 4 km southwest of Torda | 12 | 3 |
| | 6 km southwest of Torda | 4 | 2 |
| DR55 | 1 km southeast of Bašaid | — | 7 |
| DR56 | 9 km south of Kikinda | — | 1 |
| | 10 km south of Kikinda | — | 1 |
| | 6–7 km northeast of Bašaid | 6 | — |
| DR60 | Idvor (M. Dević) | — | 1 |
| DR80 | 2 km southeast of Dobrica | 1 | — |
| DR91 | 2 km southeast of Konak | 1 | — |
| DQ69 | Sakule (M. Dević) | — | 7 |
| ER20 | 3 km east of Vatin | 7 | 1 |
| EQ18 | Potporanj (J. Rašajski) | 7 | — |
| EQ19 | 3 km northwest of Vlajkovac | 42 | 16 |
| | Total | 308 | 124 |

and pastures provide necessary foraging areas for falcons during the nesting season. Red-footed falcons are probably more abundant and widely dispersed in Banat due to this reason. Conversely, Bachka supports few falcons because it is dominated by intensive row-crop agriculture.

RESUMEN.—Registré 308 y 124 parejas reproductivas de *Falco vespertinus* en Voivodina, entre 1990 y 1991, respectivamente. La mayoría de los halcones nidificaron en la región este de Banat, mientras que sólo 4–5 parejas nidificaron en Bachka, ninguna nidificó en Srem. *Falco vespertinus* ocupó nidos de colonias de *Corvus frugilegus* (>90% de las veces), *Pica pica* y *Corvus corone cornix*. En 1990, las tres colonias más grandes estaban en las vecindades de las villas de Jazovo, Melenci y Vlajkovac, las que tenían 147, 46 y 42 parejas reproductivas, respectivamente. Las mismas colonias soportaron solamente 29, 27 y 16 parejas en 1991. He atribuido esta diferencia al frío y lluvia primaveral ocurridas en 1991, las que provocaron un retardo en la nidificación de *C. frugilegus*, lo que previno la ocupación de los nidos por parte de los halcones. La falta de insectos-presa probablemente también contribuyó a la declinación en el número de parejas nidificantes de halcones.

[Traducción de Ivan Lazo]

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# ARTIFICIAL NEST STRUCTURE DESIGN AND MANAGEMENT IMPLICATIONS FOR THE LESSER KESTREL (*FALCO NAUMANNI*)

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KEY WORDS: Falco naumanni; *lesser kestrel*; *management*; *nest box*; *Spain*.

ABSTRACT.—The European population of the lesser kestrel (*Falco naumanni*) has experienced a sharp decrease in recent decades. Because they nest mainly in man-made structures, building deterioration has been an important cause of local declines when roofs collapse and nest sites become limiting. I tested two designs for artificial nest structures to be used in old buildings and a special roof tile that should increase the availability of nest sites in Spain. The first structure was made of wood and fitted under the roofs of buildings. Of 229 structures installed, 41.4% were occupied by 95 pairs of kestrels nesting in buildings. The special roof tile was tested as a nest entrance in deteriorated roofs. Of 94 tile entries installed, 23 were used by 51.1% of all pairs. The second nest structure was ceramic. Of 29 ceramic structures installed, 10 were used by breeding pairs. Although ceramic nesting structures are easy to install nearly anywhere, care must be taken to avoid locations exposed to the sun because ceramic structures can develop high internal temperatures when exposed to direct solar radiation. Both nest structures and the tile entry can be fitted to old and new buildings to prevent roof deterioration and to allow for the establishment of new colonies.

The lesser kestrel (*Falco naumanni*) is a species whose distribution has decreased dramatically in recent decades (Biber 1990). In Spain, the population decreased from about 100 000 pairs in 1960 to less than 50 000 in 1970 and only 5000 in 1988 (González and Merino 1990). Land-use changes in breeding areas are considered the main cause of the decline (Donázar et al. 1993) but lack of nesting places has also become a serious local problem. In Spain, 95% of these small and colonial falcons nest in buildings (González and Merino 1990), so restoration (closing the small holes in the walls or roofs), deterioration and the collapse of old buildings have caused several colonies to disappear (González and Merino 1990, Negro 1991, Tella et al. 1993). Use of artificial nest structures has been recommended to ease the problem caused by the loss of nest sites (Biber 1990, Blanco and González 1992). Use of these structures has been an effective management tool for European and American kestrels (*Falco tinnunculus* and *F. sparverius*) in areas with poor nest-site availability (Hamerstrom et al. 1973, Village 1983). Despite several efforts to install artificial nest structures in Spain, efforts to reestablish lesser kestrels have had only limited success.

This study tested designs for artificial nest structures that would be easy to install. Two kinds of nest structures and a special entrance tile were designed and tested in several nesting colonies of lesser kestrels.

The study took place in Monegros (Aragon) and Catalonia, Spain. In Monegros, about 98% of the kestrels nested under roof tiles in abandoned buildings. In this area, an increasing population of >200 pairs of lesser kestrels is dispersed over more than 30 colonies (Tella et al. 1993). In Catalonia, a reintroduction program was being developed (Pomarol 1993) and a few small colonies had recently been established.

One artificial nest structure was made out of wood and was fitted under the roofs of buildings (Fig. 1; González and Merino 1990). A total of 229 of these structures was tested from 1990–95. Kestrels could go under the roof tiles through cracks and holes in deteriorated tiles. From there, they entered the nest box through a hole that was bored through the reeds and mud used in the construction on roofs. The entry was approximately 40–60 cm in length and the tunnel was not straight to ensure that the bird could not see directly outside from inside the nest structure. Both characteristics are commonly found in natural, lesser kestrel roof nests. To avoid causing roof leaks, 94 special roof tiles commonly used in new buildings for roof ventilation, were tested in 1993–95 as entryways to nest structures (Fig. 2).

The second nest structure was ceramic and made for easy installation in a variety of conditions. A total of 29 of these structures was tested in 1993–95, in two roofless, ruined buildings (Fig. 3). It had a lateral entrance so females could not see directly out and entry was 8 cm in diameter. Several small holes (0.3 cm diam.) were made in the rear to increase ventilation.

Both nest-box designs and entry tiles were installed in buildings used by nesting kestrels so there was a choice between natural and artificial nest sites.

Because high temperatures can be reached inside ceramic pots exposed to the sun (Tella et al. 1994), three changes were made to ceramic nest structures to determine how the thickness and color of construction materials can affect internal temperatures that develop within these pots. In one case, the ceramic nest structure was made with thin walls (0.5 cm in thickness). In a second case the structure was made with thick walls (1.0 cm in thickness) and in the third case, the ceramic structure

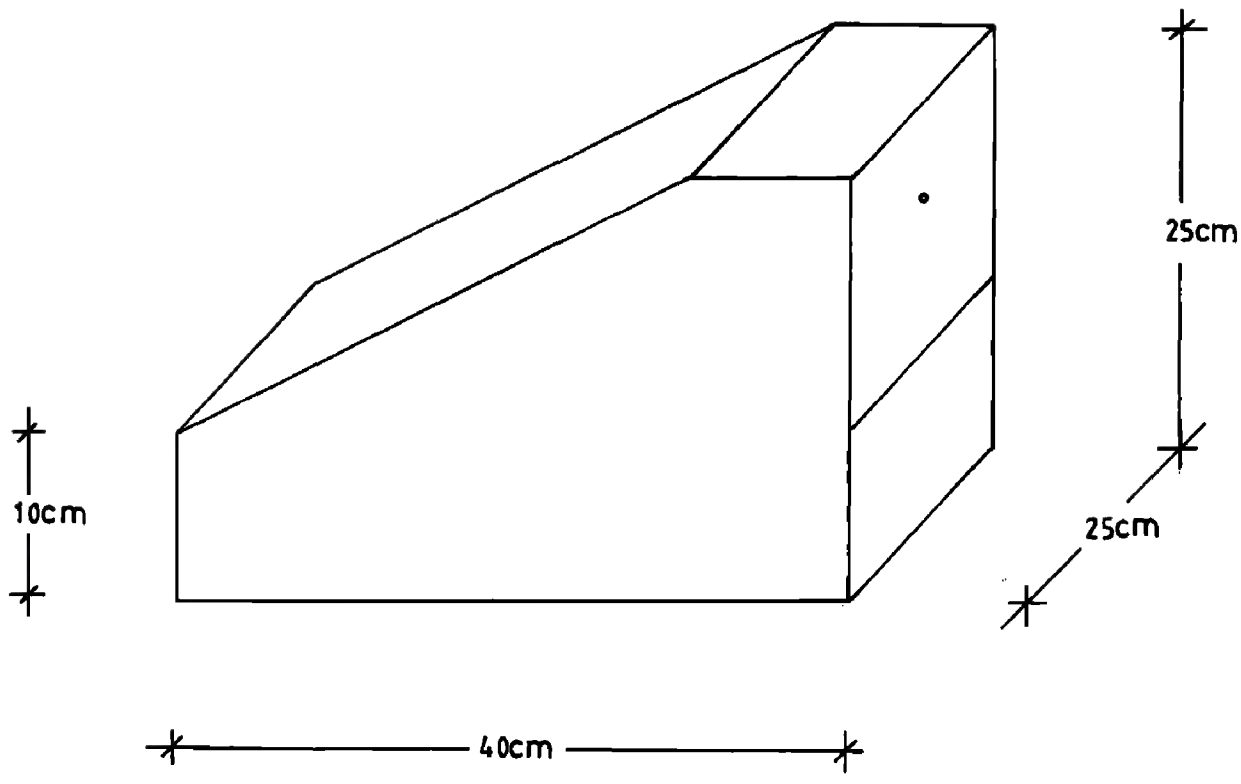


Figure 1. Wooden nest structure installed under tile roofs for nesting lesser kestrels.

was whitewashed. All three types of ceramic kestrel boxes were installed on the same roof and a maximum/minimum thermometer was placed in each. Over a 26-d period in July 1994, maximum daily temperatures were recorded inside these ceramic pots and wooden nest boxes, natural cavities under roof tiles, as well as outside in the shade. Data were analyzed using ANOVA and differences between means was determined with a LSD test.

Forty-one percent of the 229 wooden structures were occupied by 95 known breeding kestrel pairs (Table 1).

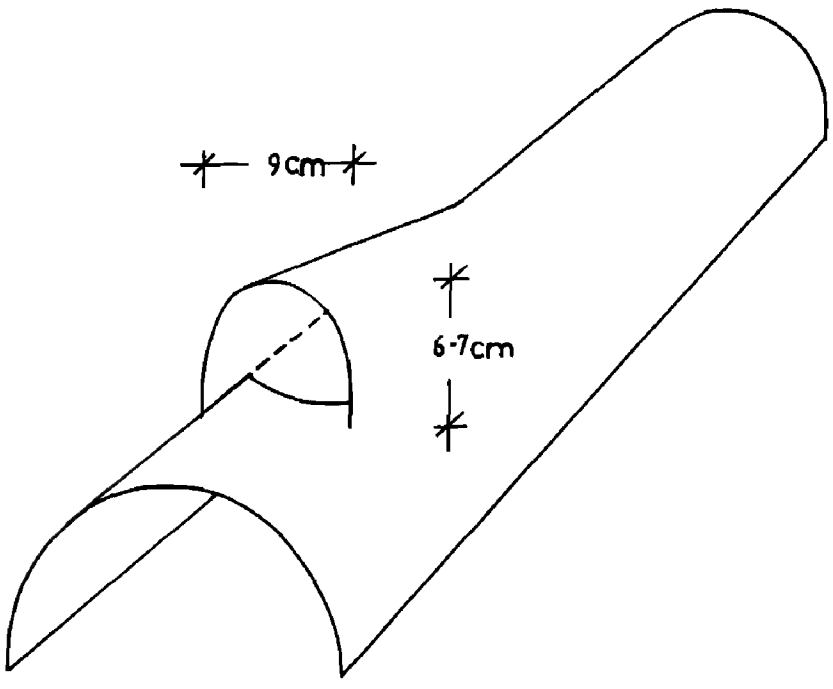


Figure 2. Special tile used as an artificial entrance for the lesser kestrel.

The spotless starling (*Sturnus unicolor*) also used these structures as did a similar species, the European starling (*S. vulgaris*), which has been found to be a regular breeder in many different designs for nest structures (Gauthier 1988). Little owls (*Athene noctua*), jackdaws (*Corvus monedula*), stock doves (*Columba oenas*), dormice (*Elyomis quercinus*) and rats (*Rattus rattus*) also used the structures sporadically.

A total of 94 tiles were fitted in the roofs of buildings used by four colonies of breeding lesser kestrels. Fifty-four were installed in combination with wooden nest structures and 40 were placed over natural cavities. Twenty-three pairs (51.1%) of the 45 known pairs nesting in these buildings chose these tiles as the entrance to their nests and starlings and little owls also used them sporadically.

Twenty-nine ceramic nest structures were located in two colonies. Ten (28.5%) of the 35 known breeding pairs in these colonies nested in the ceramic structures. The only other species to use this type of structure were the spotless starling, house sparrow (*Passer domesticus*) and scops owl (*Otus scops*).

Thin-walled ceramic nest structures developed significantly higher mean temperatures (41.3 ± 3.2°C, $P < 0.05$) than thick-walled ceramic structures (39.1 ± 3.5°C), whitewashed ceramic structures (34.9 ± 2.3°C), natural cavities (37.0 ± 2.5°C) and wooden nest boxes under roofs (33.3 ± 1.6°C). Temperatures in wooden nest boxes installed under roofs also varied less than did temperatures in ceramic structures ($P < 0.05$).

Wooden nest structures were easy to check from inside buildings minimizing disturbance to colonies. Unfortu-

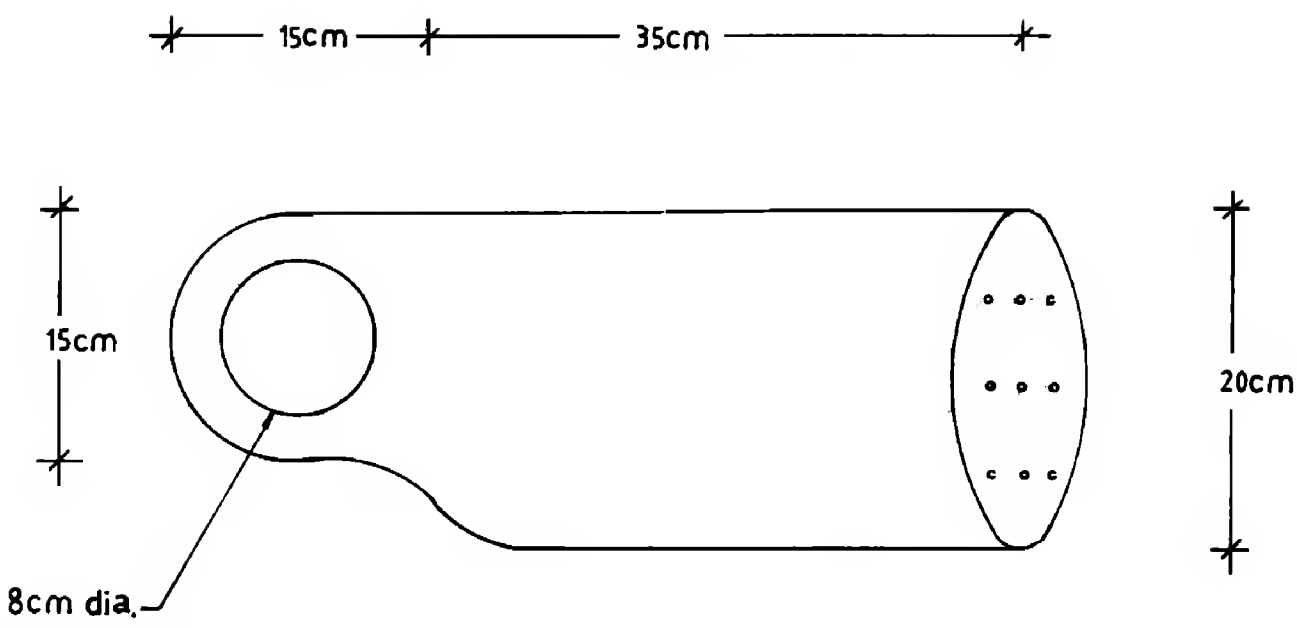


Figure 3. Ceramic nest structure used for nesting lesser kestrels.

nately, they could not be used in all buildings, particularly in ruins with no roofs or in buildings where it was not possible to reach spaces under roofs.

Different materials have been used in constructing artificial nest structures (Soulliere et al. 1992), but few designs have incorporated ceramic materials (Bernal 1991). Ceramic nest structures have the advantage of being easy to install in any building (they are simply attached with concrete) and they do not deteriorate easily. However, they have a drawback in their potential to develop high internal temperatures. Temperatures as high as 49°C was reached inside ceramic structures during this study and temperatures as high as 55°C have been reported by Tella et al. (1994). Temperature extremes are not only lethal to chicks but also eggs (Webb 1987). Varnishing ceramic structures causes even higher temperatures to be reached (Bernal 1991). My results showed that only ceramic nest structures with thick and whitewashed walls should be used in places exposed to sun.

Table 1. Use artificial nest structures by lesser kestrels in Spain.

| YEAR | # OF COLONIES | # OF NEST STRUCTURES | # OF NEST STRUCTURES OCCUPIED | % OF PAIRS NESTING IN BOXES |
|---|---|---|---|---|
| 1990 | 1 | 10 | 8 | 30.7% |
| 1991 | 4 | 48 | 19 | 21.8% |
| 1992 | 4 | 48 | 26 | 31.7% |
| 1993 | 8 | 65 | 33 | 35.8% |
| 1994 | 2 | 29 | 5 | 38.4% |
| 1995 | 2 | 29 | 4 | 36.3% |
| TOTAL | 21 | 229 | 95 | 30.5% |

To prevent further declines of the lesser kestrel in Spain, reconstruction of buildings supporting breeding colonies of lesser kestrels should be done outside the breeding season and the holes or cavities in walls of these buildings that are suitable for nesting kestrels should not be closed, as has already been proposed by González and Merino (1990) and Biber (1990). If holes must be repaired, nest structures similar to those I tested should be used, even in new buildings, to provide lesser kestrels with the opportunity to nest and recolonize previously occupied areas. Use of special roof tiles as access openings to nests also makes it possible to equip roofs with artificial cavities for lesser kestrel colonies without causing harm to buildings. A simple solution would be to subsidize the use of these tiles in new constructions in appropiate areas.

RESUMEN.—Las poblaciones de cernícalo primilla han padecido una fuerte regresión en las ultimas décadas. Debido a que esta rapaz nidifica principalmente en edificios, la escasez de lugares de nidificación motivado por las reconstrucciones o el deterioro de estos, son una causa local importante de desaparición. Dos tipos de cajas-nido y una teja especial han sido recientemente probados con éxito. El primero fue hecho de madera, y fue colocado bajo el tejado. De 229 cajas instaladas, el 41.4% fueron usadas por el 30.5% de las parejas nidificantes en esas colonias. Para evitar el deterioramiento del tejado, se probó una teja especial que sirviera de entrada al nido. De 94 tejas, 23 fueron utilizadas por el 51.1% de las parejas. La segunda caja nido fue hecha de cerámica. De 29, 10 fueron utilizadas por el 28.5% de las parejas. Aunque esta puede ser utilizada en cualquier sitio, se debe tener cuidado por las altas temperaturas que se pueden alcanzar en su interior. Todos estos nidos artificiales pueden ser colocados tanto en edificios nuevos

como viejos, evitando el deterioro de estos y favoreciendo el establecimiento de nuevas colonias.

[Traducción del Autor]

ACKNOWLEDGMENTS

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# BOOK REVIEW

EDITED BY JEFFREY S. MARKS

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**Pirate of the Plains: Adventures with Prairie Falcons in the High Desert**. By Bruce A. Haak. 1995. Hancock House, Blaine, WA. 208 pp., 74 color photos. ISBN 0–88839-320-2. Paper, $21.00.—Here is a story about the Zen of doing field research that will motivate the novice and rekindle the enthusiasm of the professional. Bruce Haak takes us back to a carefree time when pursuit of knowledge and desire to explore the unknown were the only motivations necessary to devote one's life to biological research. This book expanded my understanding of prairie falcons (*Falco mexicanus*), but more importantly made me long for camping under clear western skies and waking up with the anticipation of finding a new falcon aerie. Readers will find this book easy to relate to, humorous, laced with conservation ethics, and quick to finish.

The book is organized around Haak's formal and informal research on prairie falcons. It starts with his less-structured wanderings in eastern Oregon as an undergrad at Oregon State University. Rather than spending the late 1960s in Vietnam's rice paddies, Haak chose to further his education. He relates his experiences with the high desert climate, ghostly falcons, and open land during frequent weekend excursions from OSU. This introduction to the land and bird progresses into a more serious survey of falcons mandated by the Endangered Species Act, where Haak presents useful information on the variety of habitats exploited for nesting by prairie falcons in eastern Oregon. We next follow Haak to the lava beds of northern California to study the foraging habits of falcons for his master's thesis. Here, details on the foraging maneuvers, spacing patterns of pairs, trapping techniques, use of falconry to supplement science, and radiotracking are presented. The reader gets an excellent feeling for the high desert environment, which Haak accurately describes as a place where "the wind never stops blowing." Haak touches upon a variety of behavioral observations detailed in his thesis and gives some data and insights into subadult breeding, parental behavior at the nest (including prey caching), and ranging habits. The remainder of the book recounts observations made during less-structured field projects funded by "back pocket grants." There is a useful comparison of prairie falcon and peregrine falcon (*Falco peregrinus*) habits and interesting discussions of potentially symbiotic relationships between common ravens (*Corvus corax*) and prairie falcons. The book ends with a summary of prairie falcon population status across the region and a discussion of the positive and negative effects of humans on this species.

Haak introduces the reader to a variety of important conservation issues in the Columbia River basin. He suggests how overgrazing, coyote control, rangeland "improvement," private use of public lands, agriculture, and water diversion have shaped the land and influenced its natural inhabitants. He offers strong opinions on many of these forces that most biologists would be quick to agree with. A few of my favorites are his description of cheat grass (*Bromus tectorum*) as the "most useless plant" to invade the area, and coyote control as simply a "wasted effort." The reader familiar with this ecosystem and its abuse at the hands of humans will applaud Haak's candor during these wanderings. The reader that has not spent time in the Columbia basin will finish the book with an excellent introduction to the conflicts between human settlement and biodiversity in this harsh environment.

I found this book to be a delightful story of a magnificent land and its changes over the past three decades. The only frustration I had was over the lack of scientific citations or footnotes. Many scientific studies are referred to, but the reader has no way of knowing which articles are being discussed. There also was little to no quantification of Haak's data. Such a scientific assessment of prairie falcons was not the intent of this book, but a bit of scholarship could have been included without detracting from the book's style. Lack of scholarship is especially disappointing because these data are

available only in Haak's thesis, none of which has been published in the general literature. In summary, this is not a technical account of prairie falcon behavioral ecology, but it belongs on the shelves of raptor field biologists and should be recommended reading for young researchers intent on exploring the mysteries of vertebrate biology.—**John M. Marzluff, Sustainable Ecosystems Institute, 30 East Franklin Road, Suite 50, Meridian, ID 83642 U.S.A.**

*The Journal of Raptor Research* is distributed quarterly to all current members. Original manuscripts dealing with the biology and conservation of diurnal and nocturnal birds of prey are welcomed from throughout the world, but must be written in English. Submissions can be in the form of research articles, letters to the editor, thesis abstracts and book reviews. Contributors should submit a typewritten original and three copies to the Editor. All submissions must be typewritten and double-spaced on one side of 216 × 278 mm (8½ × 11 in.) or standard international, white, bond paper, with 25 mm (1 in.) margins. The cover page should contain a title, the author's full name(s) and address(es). Name and address should be centered on the cover page. If the current address is different, indicate this via a footnote. A short version of the title, not exceeding 35 characters, should be provided for a running head. An abstract of about 250 words should accompany all research articles on a separate page.

Tables, one to a page, should be double-spaced throughout and be assigned consecutive Arabic numerals. Collect all figure legends on a separate page. Each illustration should be centered on a single page and be no smaller than final size and no larger than twice final size. The name of the author(s) and figure number, assigned consecutively using Arabic numerals, should be pencilled on the back of each figure.

Names for birds should follow the A.O.U. Checklist of North American Birds (6th ed., 1983) or another authoritative source for other regions. Subspecific identification should be cited only when pertinent to the material presented. Metric units should be used for all measurements. Use the 24-hour clock (e.g., 0830 H and 2030 H) and "continental" dating (e.g., 1 January 1990).

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## Raptor Research Foundation, Inc., Awards

Recognition for Significant Contributions[1]

The **Dean Amadon Award** recognizes an individual who has made significant contributions in the field of systematics or distribution of raptors. Contact: **Dr. Clayton White, 161 WIDB, Department of Zoology, Brigham Young University, Provo, UT 84602 U.S.A.** Deadline August 15.

The **Tom Cade Award** recognizes an individual who has made significant advances in the area of captive propagation and reintroduction of raptors. Contact: **Dr. Brian Walton, Predatory Bird Research Group, Lower Quarry, University of California, Santa Cruz, CA 95064 U.S.A.** Deadline: August 15.

The **Fran and Frederick Hamerstrom Award** recognizes an individual who has contributed significantly to the understanding of raptor ecology and natural history. Contact: **Dr. David E. Andersen, Department of Fisheries and Wildlife, 200 Hodson Hall, 1980 Folwell Avenue, University of Minnesota, St. Paul, MN 55108 U.S.A.** Deadline: August 15.

### Recognition and Travel Assistance

The **James R. Koplin Travel Award** is given to a student who is the senior author of the paper to be presented at the meeting for which travel funds are requested. Contact: **Dr. Petra Wood, West Virginia Cooperative Fish and Wildlife Research Unit, P.O. Box 6125, Percival Hall, Room 333, Morgantown, WV 26506-6125 U.S.A.** Deadline: established for conference paper abstracts.

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### Grants[2]

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[1] Nominations should include: (1) the name, title and address of both nominee and nominator, (2) the names of three persons qualified to evaluate the nominee's scientific contribution, (3) a brief (one page) summary of the scientific contribution of the nominee.

[2] Send 5 copies of a proposal (≤5 pages) describing the applicant's background, study goals and methods, anticipated budget, and other funding.